WATER QUALITY TREATMENT AND HYDRAULIC EFFICACY OF LABORATORY AND FIELD RAIN GARDENS

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Abstract

Urbanisation leads to increases in stormwater runoff, resulting in elevated contaminant (e.g. metal, sediment, and nutrient pollutant) loads, decreased local infiltration and greater peak flow intensities. Heavy metal contaminants of concern, primarily copper (Cu), lead (Pb), and zinc (Zn), originate from a variety of sources including wear-and-tear of vehicle parts, corrosion of alloy roofs, legacy petroleum contamination, and multifarious construction practices. Different technologies have been used to mitigate stormwater runoff, ranging from traditional drainage networks fitted with concrete proprietary devices (e.g. vortex sediment separators and filters) to more environmentally integrated sustainable solutions.

Rain gardens, a type of Sustainable Urban Drainage System (SUDS) or Water Sensitive Urban Design (WSUD), are employed to control stormwater peak flows and runoff volumes and simultaneously reduce contaminant loads to neighbouring waterways through biologically-active landscaped design. Despite increases in use of rain gardens as a best management practice (BMP) to treat urban stormwater runoff, there is a dearth of knowledge about their treatment and infiltration performance worldwide. It is believed that incorporating topsoil into rain garden design is likely to improve contaminant removal efficiencies (Davis et al. 2001; ARC 2003; Fletcher et al. 2004; Carpenter and Hallam 2010), but design recommendations are not informed by performance data which is limiting. Performance data is necessary for understanding the long-term responses of bioinfiltrative treatment systems and for modelling efforts aiming to predict their mitigation behaviour (Fletcher et al. 2004).

In order to evaluate the influence of substrate composition on stormwater treatment and hydraulic effectiveness in rain gardens, mesocosm-scale (180 L, 0.17 m^2) laboratory systems were established. Saturated (constant-head) hydraulic conductivity was determined before and after contaminant (Cu, Zn, Pb and nutrients) removal experiments on three rain garden systems comprising various proportions of organic topsoil. Raw stormwater runoff from a neighbouring Christchurch city catchment was collected, characterised, and applied in the removal efficiency experiments. The system with only topsoil had the lowest saturated hydraulic conductivity (160 mm/hr initial to 164 mm/hr final) and poorest metal (Cu, Zn) removal efficiency (Cu 0.3%, Zn 60.5% and Pb 89.5%) at a ‘standard’ contaminant loading rate (Cu = 5.99 ± 0.73 µg/min, Zn = 57.89 ± 6.06 µg/min, Pb = 13.65 ± 2.80 µg/min). The

sand-only system demonstrated good metal removal (Cu 56.4%, Zn 73.5%, and Pb 81.6%) with hydraulic conductivity (up to 805 mm/hr) adequate for practical implementation (i.e. greater than the 13 mm/hr minimum requirement (ARC 2003; MDE 2009; SFPUC 2009)). Overall, total metal amounts in the effluent were <50% of influent loads for all experiments, with the exception of Cu in the topsoil-only system, whose removal was negligible (0.3%). Greater metal removal was observed when effluent pH was elevated (up to pH 7.38). The pH increase (from an initial pH of 6.23 in raw stormwater) was provided by the calcareous sand, whereas the topsoil-only system lacked an alkaline source. Consequently, organic topsoil had poorer contaminant removal due to higher dissolved metal fractions, which are more difficult to immobilise at the lower pH. The relationship between pH and dissolved fraction was highly significant (Pearson's Correlation, $p < 0.0001$, $d_f = 76$) for Cu, Zn, and Pb.

Mesocosm-scale systems were then re-established with a calcareous substrate supplement to quantify the effects of pH augmentation on contaminant removal and hydraulic efficiencies. Mussel shells, a waste product from the shell-fish industry, were employed in two different volumetric proportions. Metal removal efficiency was increased in systems with mussel shells (Cu up to 46.6%, Zn up to 80.2%, Pb up to 88.7%) compared to the topsoil-only system (Cu 27.5%, Zn 55.5%, Pb 81.0%). Larger increases in removal efficiency were seen for Cu and Zn because increases in pH from mussel shell enhanced particulate fractions, which are easier to remove in filtration systems, while Pb is mainly in the particulate form at influent pH (Morrison et al. 1990). Effluent from systems with mussel shells also had higher hardness (hardness up to 101.7 mg/L as $CaCO_3$) compared with 22.4 mg/L as $CaCO_3$ in topsoil-only effluent. Hardness reduces metal ecotoxicity (Hyne et al. 2005). Results of these experiments show that mussel shells are a promising rain garden substrate capable of increasing metal removal efficiency and also decreasing metal ecotoxicity in effluent of bioinfiltration systems.

Concurrently, an operational field-scale "rain garden" (42 m^3; 60 m^2) in Christchurch was monitored for hydraulic throughput and contaminant removal. The field system performed extremely well at mitigating peak flows, detaining water throughout storm events and removing total suspended solids (TSS) (90.6% average removal). However, the system failed to reduce effluent median total metal concentrations (Cu = 15.9 μg/L, Zn = 139.6 μg/L, Pb =11.7 μg/L) below relevant ANZECC guidelines (Cu = 1.8 μg/L, Zn = 15.0 μg/L, Pb =5.6 μg/L) highlighting the opportunity to optimise these field designs to improve metal removal.

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Terminology

Abbreviations and Acronyms

ANZECC	Australian and New Zealand Environment and Conservation Council
ARC	Auckland Regional Council
BMP	Best Management Practice
$CaCO_3$	Calcium Carbonate
CCC	Christchurch City Council
CEC	Cation exchange capacity
COD	Chemical Oxygen Demand
Cu	Copper
DO	Dissolved Oxygen
DOC	Dissolved Organic Carbon
ECAN	Environment Canterbury
EDTA	Ethylenediaminetetraacetic Acid
EMC	Event Mean Concentration
Eqn	Equation
Fe	Iron
GFC	Glass Fibre Composite
HDPE	High-Density Polyethylene
Hill Labs	RJ Hill Laboratories
HydroEco	Hydrological and Ecological Engineering Research Group
IANZ	International Accreditation New Zealand
ICP-MS	Inductively Coupled Plasma Mass Spectrometry
ISQG	Interim Sediment Quality Guidance
LID	Low Impact Development
Mg	Magnesium
NIWA	National Institute of Water and Atmospheric Research
NO_3	Nitrate
NZ	New Zealand
NZMOH	New Zealand Ministry of Health
PAHs	Polycyclic Aromatic Hydrocarbons
Pb	Lead
PVC	Polyvinyl Chloride
QA/QC	Quality Assurance/Quality Control
RCRA	Resource Conservation and Recovery Act
RMA	Resource Management Act
SCS	Soil Conservation Service

SUDS	Sustainable Urban Drainage System
T_c	Time of Concentration
TCLP	Toxicity Characteristic Leaching Procedure
TP10	Technical Publication 10
TN	Total Nitrogen
TKN	Total Kjeldahl Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids
UC	University of Canterbury
USDA	United States Department of Agriculture
WWDG	Waterways, Wetlands, and Drainage Guide
WQV	Water Quality Volume
WSUD	Water Sensitive Urban Design
Zn	Zinc

Units

cfu	colony forming units
d	days
g	grams
hr	hours
in	inches
kg	kilograms
L	litres
m	meters
me	milliequivalents
mg	milligrams
min	minutes
mL	millilitres
NTU	nephelometric turbidity unit
SU	standard pH units
µg	micrograms

Research Outputs

Journal Articles

Good, J.F., O'Sullivan, A.D., Wicke, D., and Cochrane, T.A., (2011). Contaminant Removal and Hydraulic Conductivity of Laboratory Rain Garden Systems for Stormwater Treatment. Water Science and Technology. (in press)

Peer-Reviewed Conference Proceedings

Good, J.F., O'Sullivan, A.D., Wicke, D., and Cochrane, T.A., (2011). Appreciating Drainage Assets in New Zealand Cities: Rain Garden Treatment and Hydraulic Performance. International Water Association- Cities of the Future Conference, 22-25 May 2011, Stockholm, Sweden.

Conference Extended Abstracts and Presentations

Good, J.F., O'Sullivan, A.D., Wicke, D., and Cochrane, T.A., (2011). The Importance of pH in Ecological Stormwater Treatment Systems. New Zealand Hydrological Society Conference, 5-9 December 2011, Wellington, New Zealand. (accepted as extended abstract with intention of presenting)

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Good, J.F., O'Sullivan, A.D., Wicke, D., and Cochrane, T.A., (2010). Biofiltration Stormwater Systems to Mitigate Water Resource Contamination: Rain Gardens in Christchurch. New Zealand Hydrological Society Conference, Dunedin, New Zealand, 7-10 December 2010, 46-47 pp.

Chapter 1: Introduction

Urban areas, characterised by increased population densities, extensive imperviousness and a wide variety of industry, contribute a range of contaminants resulting from activities such as building renovation, excavations, road construction, fire, and traffic and well as wear and tear of roof and road surfaces (Zanders 2005; Jartun and Pettersen 2010). Urbanization causes an increase in stormwater runoff resulting in pollutant loads, decreased infiltration and greater peak flow intensities impacting on local and downstream aquatic ecosystems (Beasley and Kneale 2002).

1.1 Stormwater Character and Contaminant Sources

Stormwater is recognized as a substantial source of pollutants to receiving waterways (Davis et al. 2001) and increasing attention has been placed on the need to manage this stormwater (Fletcher et al. 2004). Heavy metals, a common constituent in urban stormwater, are of particular importance because, unlike organic compounds, metal elements are not degraded in the environment (Sansalone and Buchberger 1997). Heavy metals are commonly defined as any metal with a specific gravity of 5 or higher (Lapedes 1974) and are a major concern in stormwater discharges in New Zealand due to the stringent guidelines to protect the ecological health of receiving waterways. The primary stormwater heavy metals of concern in New Zealand are copper (Cu), lead (Pb) and zinc (Zn), which are ubiquitously present (Zanders 2005; Davis and Birch 2009). Local (Christchurch) urban stormwater runoff has received increasing attention in recent years, advanced by research identifying heavy metal concentrations (especially Zn and Cu) consistently above the most lenient 80% levels of protection of aquatic species defined in the contextual ANZECC guidelines (Wicke et al. 2009; Wicke et al. 2010).

Heavy metal contamination sources in urban environments include vapour from vehicle emissions, dust from vehicle tyres and brake linings, and city fires (Purves 1967; Zanders 2005; Jartun and Pettersen 2010). Table 1-1 shows anthropogenic contaminants found in stormwater and their sources. Contaminant concentrations fluctuate based on traffic load, antecedent dry days, rainfall intensity and volume, and impermeable surface characteristics (Hall and Anderson 1988). Loads can be expected to be highest on busy roads carrying a high proportion of heavy traffic, in braking and acceleration zones, where road runoff is ‘untreated’

on its way to the point of discharge (Moores and Pattinson 2008). Stormwater runoff from highways is typically more ecotoxic than runoff from urban areas (Marsalek et al. 1999), although ecotoxicity of urban stormwater is still substantial (Engstrom 2004).

Table 1-1- Sources of anthropogenic constituents in urban pavement runoff (modified from Sansalone and Buchberger (1997)).

	Brakes	Tyres	Frame and Body	Fuels and Oil	Concrete Pavement	Asphalt Pavement	De-icing Salts	Litter
Cadmium	■	■						
Chromium		■						
Copper	■	■						
Iron		■	■					■
Lead	■	■		■			■	
Nickel		■						
Vanadium				■				
Zinc	■	■	■					
Chlorides							■	
Organic Solids						■		■
Inorganic Solids			■		■	■		■
PAHs				■		■		
Phenols						■		

Roofs typically constitute a significant portion of the impervious surfaces in an urban catchment and are a major contributor to stormwater contamination (Karlen et al. 2002; Shedden et al. 2007; Pennington and Webster-Brown 2008; Egodawatta et al. 2009). The runoff from roofs, gutters and downspouts is often contaminated with zinc and copper, both as a result of atmospheric deposition and dissolution of the galvanized metal surface (Good 1993; Mason et al. 1999; Michels et al. 2002; Pennington and Webster-Brown 2008). International studies have found roof heavy metal concentrations (sum of roof deposited contaminants and roof dissolution) ranging from 20 to 5,400 µg/L Cu, 8 to 7,690 µg/L Zn, and 2.6 to 38,439 µg/L Pb (Mason et al. 1999; Karlen et al. 2002; Robert-Sainte et al. 2009). Similar dissolution studies in New Zealand (Shedden et al. 2007; Pennington and Webster-Brown 2008), including on-going research led by the Hydrological and Ecological Engineering (HydroEco) Research Group within the Civil and Natural Resources Engineering

Department at the University of Canterbury, confirm leaching of copper and zinc from roof panels, gutters and down pipes, with copper levels exceeding the 90% ANZECC guideline value up to 3,200 fold at > 4,000 µg/L (O'Sullivan et al. 2011). ARC (2005) combined data from three different studies to produce mass budgets for Zn, Cu and Pb loads in urban stormwater. Results identified urban and commercial catchment roof runoff accounted for almost all the Zn in the catchment stormwater (ARC 2005). An extensive study in France assessed the dissolution from 12 different metallic roofing materials over 14 months and concluded that more metallic species at higher concentrations were released from zinc, copper and lead roofs than from aluminium and stainless steel roofs (Robert-Sainte et al. 2009). These findings are particularly relevant to New Zealand, where copper roofing materials are increasingly popular on domestic dwellings as architectural features (Pennington and Webster-Brown 2008).

While it is generally agreed from an ecotoxicology perspective that heavy metals are those that cause harm to the environment, some metals (e.g. Cu and Zn) are essential for life in limited amounts and are vital components of enzymes, proteins and structural elements of certain organisms (Depledge and Rainbow 1990). Previously, emphasis was placed on measuring heavy metal concentrations in biota as a method of determining threat posed to humans from the ingestion of excessive heavy metal loads in edible species; however, the focus now ascertains effects of heavy metals on ecosystem populations and communities (Depledge et al. 2009). Despite the emphasis placed on contaminant concentrations, it is no longer scientifically justified to base assessment of the fate and biological effects of metals in the environment solely on their concentration alone (Landner and Reuther 2004). Metal contaminants are either in dissolved forms or bound to particulate matter, with the chemical form governed by complex interactions between variables including their concentration, pH, water hardness, hydrology, presence and concentration of other metal ions and organic ligands (Grassi et al. 2000; Beasley and Kneale 2002; Gnecco et al. 2008). Metal speciation favours the dissolved state at pH less than about pH 7, as particulate bound metals are released as free metal ions (Dempsey et al. 1993; Engstrom 2004). Ecological toxicity is highly dependent on metal speciation with dissolved metals (defined as anything passing through a 0.45 µm filter) being more bioavailable than particulate fractions (Sansalone and Buchberger 1997). For example, a study investigating the effects of water chemistry on acute ecotoxicity confirmed that Cu toxicity decreased as pH increased, suggesting that biological

species are more sensitive to the free cupric ion, which is dominant at an acidic pH (Hyne et al. 2005).

Copper

Copper is found in varying concentrations in stormwater based on inputs from residential sources including copper roofs, guttering, piping, and wiring and is also sourced from automobile brakes and tyres. Copper is one of the heavy metals that is most difficult to treat due to its ability to change between the dissolved and particulate phase at roughly the pH of urban stormwater (Pennington and Webster-Brown 2008). At this pH, copper fluctuates widely between the dissolved and particulate species making it difficult to remove and immobilize. The dissolved Cu^{2+} ions can forms strong complexes with hydroxide and organic ligands, e.g. humic and fulvic acids (Landner and Reuther 2004), thus making it difficult to remove these dissolved organic complexes from solution (Mason et al. 1999). It has, however, been shown that water-soluble Cu is not an appropriate measure for bioavailable Cu as most of it occurs as non-bioavailable organic complexes (i.e. toxicity is correlated with the concentration of free metal ions rather than total dissolved metal concentration (Depledge et al. 2009).

Zinc

Although Zn is ubiquitous in nature, 96% of its release into the global environment results from anthropogenic activities (Landner and Reuther 2004). This includes industrial activities such as electroplating, smelting and ore processing as well as erosion from agricultural land where Zn concentrations can be high in the soil (Landner and Reuther 2004). Zinc contributions in stormwater from soil erosion is less of a concern in New Zealand where soils are deficient in Zn (Robson 1993). Zinc is also sourced from corrosion of metal alloys including roofs, automobile brakes, tyres, and automobile bodies and frames (Sansalone and Buchberger 1997; ARC 2005). In general, Zn is increasingly released from sediments into dissolved species at low pH (Landner and Reuther 2004).

Lead

Lead was common in petrol and paint in New Zealand until around the 1970s and because of residual paint and oil surfaces, lead is still being conveyed to stormwater through corrosion (Davis and McCuen 2005) and leaching from roadside soils. Car brakes and tyres are also a source of lead contamination (Sansalone and Buchberger 1997). Lead is most commonly

associated with particles (Bryan and Langston 1992) and thus more efficiently removed from stormwater through filtration systems (Fletcher et al. 2004).

Nutrients

While stormwater treatment systems in New Zealand and Australia are designed to remove total suspended solids (TSS) and associated heavy metals to low levels due to stringent ecotoxicological guidelines (ANZECC 2000; ECAN 2011), many treatment systems overseas are focused on removing nutrients to address eutrophication problems (Walker 1995). Nutrients are extremely important in aquatic ecosystems as they provide components for photosynthesis – the basis of the food chain in ecosystems. However, excess nutrients (primarily nitrogen and phosphorus) can result in poor stream health, eutrophication and eventually hypoxia. Excess nutrients from cultural eutrophication promote algal growth. This algal growth dies over winter resulting in high biochemical oxygen demand (BOD) due to bacterial breakdown of the algae. Consequently, surrounding waters become oxygen-depleted leading to fish kills and anaerobic conditions that can induce biogeochemical changes in the sediments.

Nitrates are an issue in the Christchurch urban area because of horticultural activities (e.g. Marshland) and fertiliser manufacture (Ravensdown) stormwater discharge. However, in an urban catchment, nutrients are less of a problem than rural agricultural areas where nitrate and phosphate are present in excess due to incomplete uptake of nutrients from fertilizer applications and discharge from animal effluent. Despite relatively low levels of nitrate in urban environments as compared with rural (i.e. agricultural) applications, it was anticipated that nitrate might be of concern in stormwater as reported elsewhere (Taylor et al. 2005; Henderson et al. 2007). Alternate wet/dry cycles can affect nitrate leaching (Randall and Mulla 2001) and rain gardens undergo such cycles due to the sporadic nature of stormwater. If vegetation, which has been shown to facilitate nutrient retention in stormwater bioinfiltration systems (Read et al. 2008), is not maintained, nitrate-nitrogen may be easily mobilized.

1.2 Contaminant Transport

Urban contaminant transport and subsequent ecosystem degradation is directly related to impervious area (Walsh 2000). Impervious areas are commonly as low as 20% in residential areas and as much as 85% in commercial and urban areas (Novotny and Olem 1994). Increased impervious area reduces the capacity to infiltrate stormwater and thus a greater

percentage of stormwater is conveyed through the local catchment, leading to increased flows resulting in increased flooding, stream bank erosion and consequently physical disturbance of aquatic habitats (Hatt et al. 2008). Increases in impervious area due to urbanization results in decreased infiltration, reduced groundwater recharge, and decreased base flows in streams (Ferguson and Suckling 1990; Palhegyi 2010).

Figure 1-1 shows the effect of catchment imperviousness on mean annual flooding and environmental biotic index, a qualitative measure of environmental health (Walsh 2000). Stormwater discharge ratio (post-/pre- urban development) is highest in 100% drained (piped) systems at increased imperviousness (Figure 1-1(A)). Additionally, a more rapid decrease in biotic health is predicted in intensely drained catchments (i.e. depicted by line I in Figure 1-1(B)). Biotic health can be maintained at a higher level of catchment imperviousness (i.e. if catchments employ BMP drainage systems (i.e. depicted by line III in Figure 1-1(B))). These figures highlight the impact of unmitigated stormwater runoff on ecosystem health resulting from increased flow and contaminant loading rates to the receiving waterway.

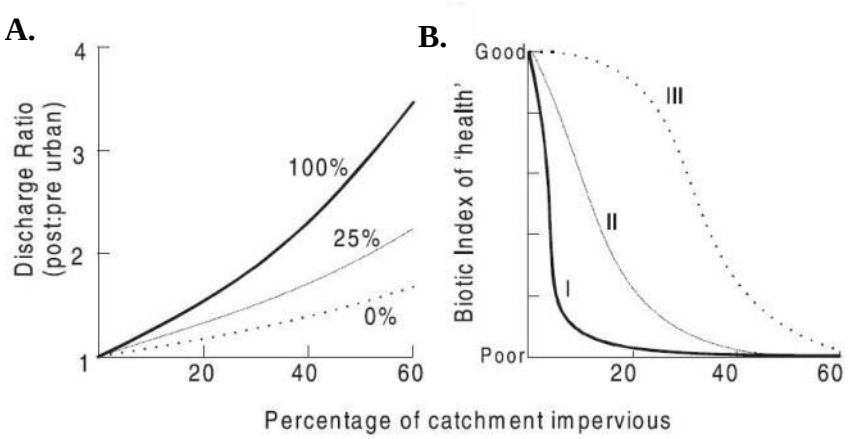

Figure 1-1- Effects of stormwater drainage regimes (modified from Walsh (2000)). Note: A) The three lines indicate predicted relationships for catchments in which 0%, 25% and 100% of the impervious area is served by stormwater drainage. (B) Hypothetical relationship between catchment imperviousness and biotic indices of stream 'health' (e.g. multimetric biotic indices or observed/ expected scores for multivariate models of community composition). The three lines indicate predicted relationships for catchments of different drainage intensity analogous to those in (A). (I) intensively drained urban areas, (II) drainage systems of moderate intensity, (III) best management practice (BMP) drainage systems.

1.3 Stormwater Environmental Regulation

A general presentation of the legislative structure is needed in order to contextualize stormwater treatment and understand how treatment designs are currently influenced and driven by legislation. In New Zealand, the Resource Management Act (RMA), which was passed in 1991, is the main piece of legislation that strives to "prevent, reduce and mitigate" adverse environmental effects (Harris 2004). The Australian and New Zealand Environment and Conservation Council (ANZECC) has developed the ANZECC guidelines. These provide an authoritative guide for setting water quality objectives required to sustain current, or likely future, environmental values for natural and semi-natural water resources in Australia and New Zealand (ANZECC 2000). They provide an effects-based guidance, which evaluates a wide range of contaminants, including those in stormwater, and provide threshold values to protect water quality (defined as percentage of species in receiving waterway not to be affected) in different aquatic ecosystems. A 90% species protection level is commonly applied to urban stormwater in Christchurch. While the ANZECC guidelines are not a legal statute, they are typically adopted in the local consenting process that ultimately becomes legally binding. The water quality objectives (detailed in Section 1.3.2.1) of stormwater treatment devices must incorporate the ANZECC 90% guidelines prior to discharging into neighbouring surface water bodies (ECAN 2011). Resource consents governing these discharges are typically enforced at the edge of the mixing zone (< 200 m along longest axis of the river for continuously flowing water and <20 m for intermittent flow), where the discharge meets the surface water body (ECAN 2011).

In addition to the RMA and ANZECC guidelines, Environment Canterbury issued the Natural Resources Regional Plan and the Christchurch City Council has developed a design guide, titled Waterways, Wetlands and Drainage Guide (WWDG). These provide further guidance on environmental control which includes managing stormwater runoff in Christchurch. Figure 1-2 (extracted from the Canterbury Natural Resources Regional Plan 2009) represents the overall legal hierarchy in New Zealand by which water managers are required to abide when managing stormwater in New Zealand. The top level of the hierarchy, the RMA, establishes ultimate national objectives for environmental management, which become increasingly detailed at the local level, ultimately resulting in legally enforceable resource consents (or permits) for stormwater activities.

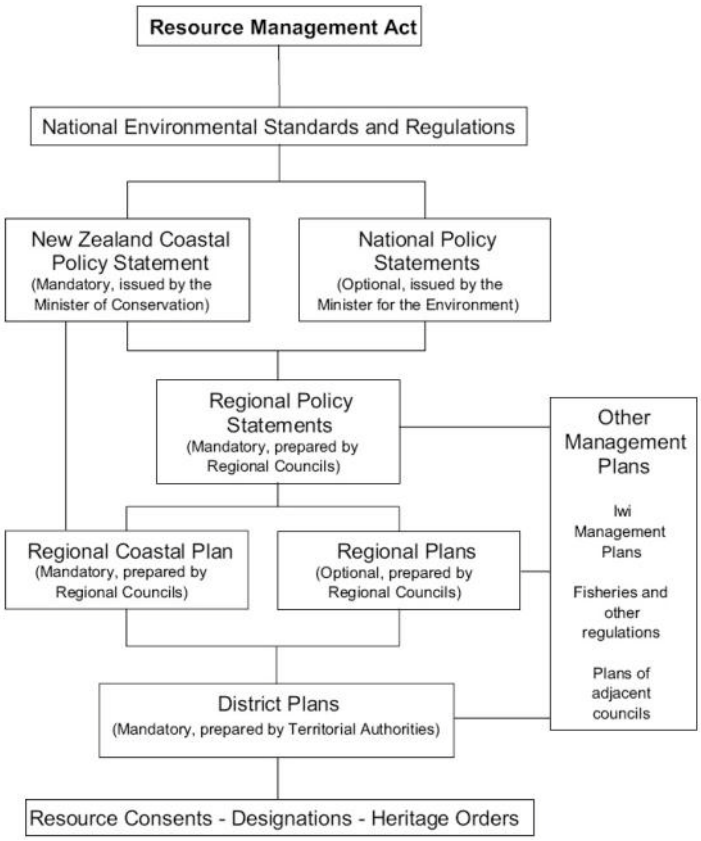

Figure 1-2- Environmental legislative hierarchy in New Zealand (ECAN 2009)

Stormwater is typically perceived as non-point source (or diffuse) pollution nationally and the point-sources contributing to stormwater contamination (e.g. discharge points of road, roof and pavement runoff) are governed by a set of laws, standards, and policies with respect to environmental protection (Figure 1-2). The ANZECC guidelines aim to protect the receiving waterway ecosystem since stormwater in New Zealand is typically discharged to neighbouring surface water bodies rather than piped to a treatment facility. While stormwater runoff in older suburbs drains directly into the neighbouring waterway, newer (and some retrofitted older) developments are typically required through resource consent to demonstrate effective mitigation of the stormwater contaminants, through passive infiltration systems (e.g. a rain garden) or manufactured propriety devices (e.g. Hynds filter systems and vortex separators). Most of these methods rely on suspended solids removal by filtration to achieve compliance with the ANZECC guidelines. In some newer residential communities, stormwater is eventually conveyed to shallow groundwater recharge following on-site infiltration in the form of detention ponds and other similar infiltration designs.

1.4 Stormwater Mitigation

Chemical contaminants from roofs and roads in highly urbanised areas are susceptible to leaching due to weather, erosion and natural decay of these surfaces (Jartun and Pettersen 2010). Stormwater is a major pathway for detaching and transporting urban contaminants into the natural environment and continues to be a leading cause of degradation of urban waterways (Walsh 2000; Egodawatta et al. 2009). For this reason, some New Zealand government authorities have focused efforts to educate local communities on the negative impacts of stormwater. Moreover, they have moved from a conveyance off-site approach to a more on-site infiltrative treatment approach in mitigating stormwater contamination of local waterways (CCC 2003).

Stormwater management should optimize three treatment goals: to minimize total stormwater volume, to mitigate peak stormwater flows, and to remove environmental contaminants. Different systems have been used to treat stormwater runoff in New Zealand and internationally including traditional pipe drained networks fitted with concrete proprietary ("off the shelf") devices, large detention systems, as well as more "ecologically" integrated systems with each type focussing different emphasis on the three defined stormwater management goals.

1.4.1 Traditional Urban Drainage

Traditionally, urban stormwater was quickly transported underground to avoid potential flooding hazards (Stephenson 1981). In larger cities overseas until around the 1960s, stormwater was transported through combined sewer systems to wastewater treatment plants (Stoddard et al. 2002). These combined systems were sized to serve large storm events (20 to 200 times the size required for sanitary wastewater systems); however, with increasing impervious area as a result of urbanisation and larger return period storms, the systems are subject to combined sewer overflows, which discharges dilute untreated sewage to natural waterways (Camp 1966). These discharges are detrimental to recreational activities, catastrophic to local ecosystems and also bring negative press and subsequent pressure from the community to reduce combined sewer overflows. In an effort to reduce these overflows, the systems were continually expanded to achieve appropriate capacity. To put the size of some of the larger systems into perspective, the current system installed in Chicago (USA), serving 5.5 million people across more than 2,200 km^2, consists of over a hundred miles of

tunnels from 3 to 10 meters in diameter and reservoirs with more than 70 million cubic meters of capacity (Theriot and Tzoumis 2005).

Around the 1960s, new single source stormwater systems were installed separate from the sanitary waste system; however it was not feasible (and cost-prohibitive) in many older cities to disconnect the stormwater from the combined sewer system. Estimates made around 1959 to separate sanitary and stormwater systems for Chicago indicated a cost of about $17,000 USD/acre ($42,000/ha), which for 135,000 acres (54,500 ha), amounted to approximately $2.25 billion USD (Camp 1966).

In the USA, the Clean Water Act (1972) and the National Pollutant Discharge Elimination System (NPDES) were established to control water pollution by regulating point sources that discharge pollutants into water bodies. These regulations were main drivers to introduce stormwater management initially for flood control and peak flow management, and then later for water quality treatment. New separate stormwater systems were established and consisted of piped networks with propriety treatment devices and sumps to meet minimum treatment goals, which were established to reduce contaminant loads entering the natural environment. While these single source (i.e. stormwater) piped systems do not have combined sewer overflows, they are still subject to managing increasing volumes of stormwater due to urbanization and associated increases in impervious area. As the systems expand, more concrete pipes, detention basins and proprietary treatment devices (i.e. vortex separators, sedimentation sumps, filters, etc.) are required. Both combined sewers and single source stormwater piped networks reduce peak flows through integrated detention basins, and potentially remove some environmental contaminants through sedimentation or filtration system. However, these piped systems are unable to reduce total stormwater volume.

1.4.2 Low Impact Development

The term Best Management Practices (BMPs) was adopted in the United States for stormwater management techniques including detention and retention basins, constructed wetlands and sand filters. In the early 1990s, Low Impact Development (LID) was introduced and piloted in Prince George's County, Maryland (PGC 1993; Carpenter and Hallam 2010) as a way to mitigate urban runoff and included bioretention, green roofs, and permeable pavement. Subsequently, similar management practices were developed around the world

including Sustainable Urban Drainage Systems (SUDS) in Europe and Water Sensitive Urban Design (WSUD) in Australia to tackle water-focused sustainable development. One of the primary goals of the approach is to integrate water cycle management into urban planning, using water as a resource to be conserved. Due to extreme drought beginning around the turn of the century, Australia has pioneered WSUDs underpinned by key principles in sustainability including water recycling, waste minimisation and environmental protection (MW 2005). Stormwater that was historically stored in concrete detention basins and/or discharged without treatment to surface waters is being treated more naturally through infiltration and used to recharge groundwater and replenish base flows in streams to promote environmental flows (Roy et al. 2008). These approaches help to reduce both the peak and total volumes of stormwater and also serve to improve waterway health by providing some level of water treatment.

1.4.2.1 New Zealand Stormwater Treatment

In New Zealand, the primary guidance for engineered stormwater treatment designs is provided in the Auckland Regional Council (ARC) Technical Publication 10 (TP10) (ARC 2003). Other stormwater design standards are available in local manuals including the Christchurch City Council's *Waterways, Wetlands and Drainage Guide* (CCC 2003); however, no design criteria for rain gardens are specified, and engineers are directed to the TP10 for design details. The TP10 uses an "effects based – best practical option" approach to determine treatment targets, including 75% removal of Total Suspended Solids (TSS) in stormwater treatment devices. In Auckland, this is interpreted on a long-term average basis but in Christchurch, it applies to each storm event (Smythe et al. 2007). The 75% TSS removal level originates from a study under Auckland conditions that found that treating 80% of all the runoff (i.e. water quality volume), resulting in 75% TSS removal, was at the marginal point of return, i.e. whilst it is possible to engineer systems to remove >75% TSS, it can be very costly for the added benefit (Smythe et al. 2007). By removing TSS, it is assumed that other contaminants of concern such as particulate trace metals, particulate nutrients, and oil, grease and bacteria attached to sediments are concurrently removed (Smythe et al. 2007). However, this assumption is currently debated amongst the engineering profession and water quality scientists since metals can prevail in the dissolved form (Lee and Jones-Lee 1993).

In 2003, Christchurch City (population 369,000), New Zealand underwent a paradigm shift in urban water management towards implementing ecologically integrated drainage infrastructure through recognising six values, namely: culture, heritage, ecology, recreation, landscape and drainage in new and retrofitted urban developments (Figure 1-3). The Christchurch Council's aim to replace traditional piped structures, which incur inevitable maintenance and offer minimal benefits besides drainage, with natural treatment systems (CCC 2003; ECAN 2009). The Auckland region (population 1.3 million) is adopting a similar approach by spending >NZ $5 billion over the next 10 years to replace deteriorating pipe networks with natural low-impact (i.e. rain garden) designs to service all water and wastewater (including stormwater) demands from new developments (Pandey et al. 2005). Auckland Regional Council (now Auckland Council) estimated they would save approximately $5 million/yr from this approach with potential for increased savings in the longer term (Pandey et al. 2005) highlighting recognition of the appreciating value of natural treatment systems (e.g. Figure 1-3). These ecological systems control peak flow, stormwater volumes and can simultaneously reduce contaminant runoff to neighbouring waterways. They are integrated into the catchment by conveying stormwater runoff through their biologically-active landscaped design before infiltrating to groundwater or discharging to surface waters.

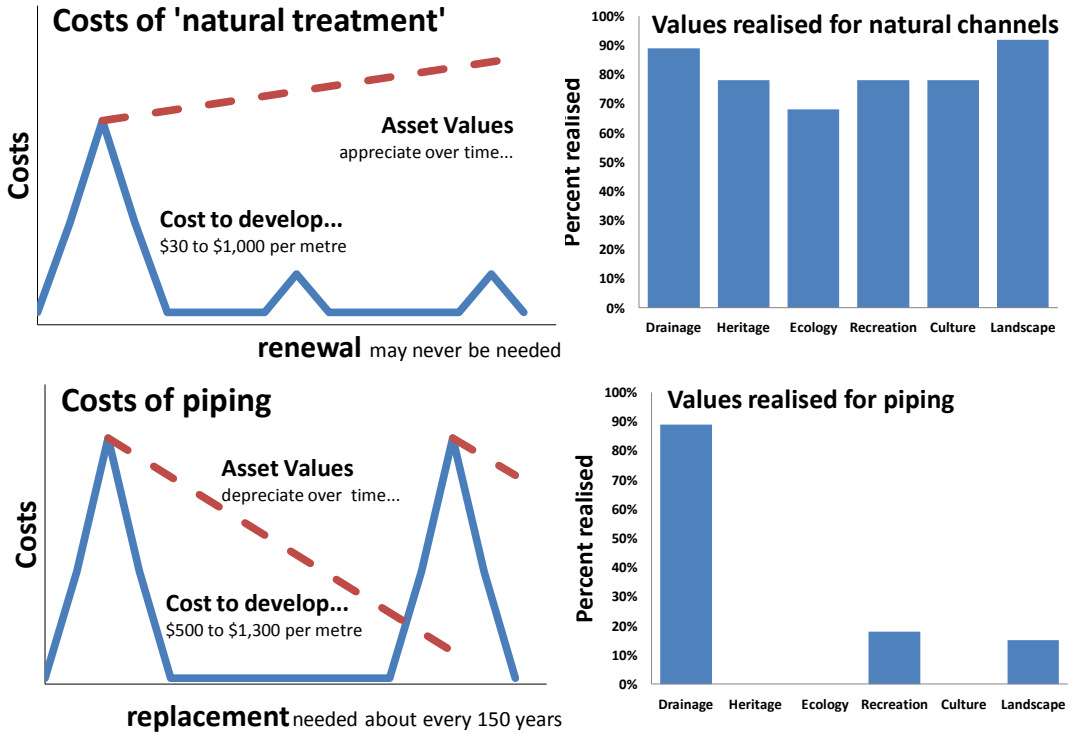

Figure 1-3- Comparison of piping and natural waterway treatment (modified from CCC (2003))

1.4.2.2 Rain Gardens

Rain Garden Designs

Rain gardens, a 'biologically active' treatment design, are characteristically shallow depressions integrated in the landscape that are planted with trees and/or shrubs, and covered with a bark mulch layer or organic ground cover overlying a gravel infiltration layer (Dietz and Clausen 2005). Most rain gardens comprise an inflow zone or pipe conveying stormwater runoff and some include an under drain channel exiting to the neighbouring waterway after infiltrating through the system. A schematic of a New Zealand rain garden design is presented in Figure 1-4, in which water flows overland as sheet flow (or can be routed via a pipe network underground to the rain garden). It is desirable to have a pre-treatment buffer (silt trap) or sump to remove large sediment and reduce the potential for long-term clogging of the system. Stormwater infiltrates vertically downwards through the ground cover (mulch), planting soil, sand and ultimately infiltrates to groundwater or is captured in an under drain system which discharges to a surface water body due to poorly infiltrating soils. The organic component of the rain garden (i.e. vegetation, mulch and topsoil) is thought to play a key role in the removal of contaminants and enhance pollutant removal, particularly for metals (Davis et al. 2001; ARC 2003; Fletcher et al. 2004).

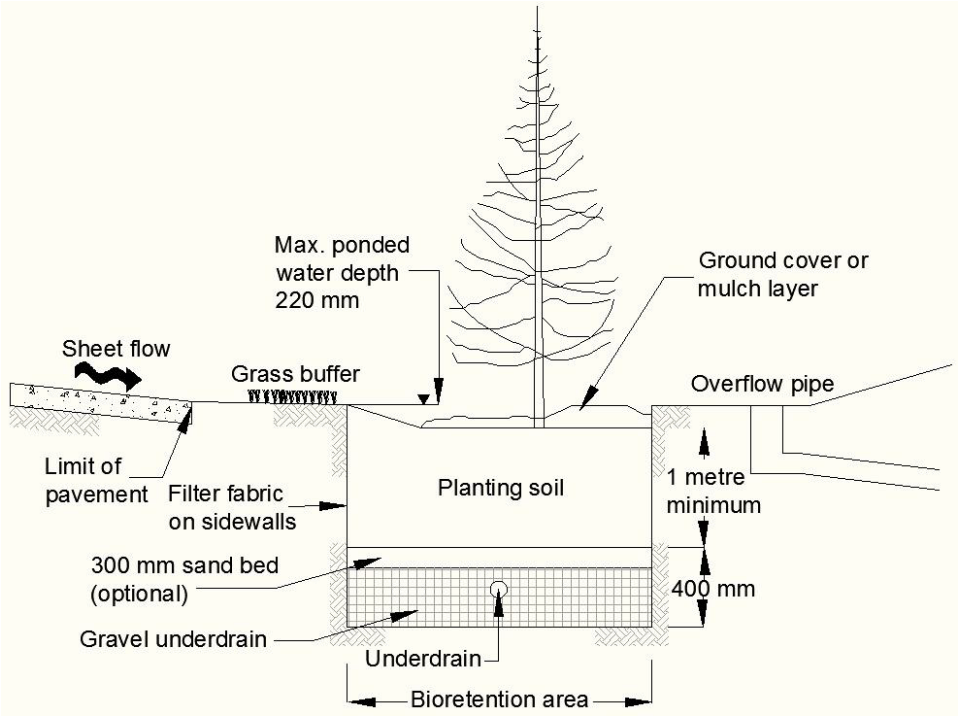

Figure 1-4- Typical rain garden design (modified from ARC (2003)

Rain gardens allow stormwater to infiltrate, recharge aquifers, and reduce peak flows. In addition, they are expected to provide water quality treatment via several processes including adsorption, decomposition, ion exchange, and volatilization (PGC 1993; Dietz and Clausen 2005). Rain garden filtration substrate should therefore be selected to target specific pollutant removal without undermining hydraulic throughput whilst maintaining localised flood control (Fletcher et al. 2004). Their designs must consider and balance both water quantity and water quality objectives.

Rain Garden Performance

Laboratory studies to investigate stormwater treatment media in New Zealand performed by Pandey et al. (2003) observed removal of Cu, Pb and Zn to be highest from filter media containing lime and bottom ash (removing over 90% of each of the metals), whilst a mixture of Sphagnum and bottom ash removed >94% of metals. This research was designed for permeable reactive barriers; however laboratory design setup was very similar to rain garden design. The laboratory results were verified in the field although flow data was not supplied and thus accurate contaminant loading and removal rate efficiencies could not be determined. Rate removal efficiencies are important in optimizing design sizes for stormwater treatment systems. Also, the study did not differentiate between dissolved and particulate metals, but did speculate that higher removal efficiencies resulted from lower metal solubility as pH increased.

Laboratory-scale bioretention studies carried out by Davis et al. (2001) in College Park, Maryland (USA) on urban runoff reported high reductions in metals (> 90% removal of copper, lead and zinc) with approximately 80% reduction in total phosphorus, 65-75% reduction in total kjeldahl nitrogen (TKN) and 60-80% reduction in ammonium. However, they applied synthetic stormwater to the systems, which is unlikely to mimic the complexities of natural stormwater. Although their study provided interesting information on bioretention removal efficiencies for one pilot-scale design type, it did not investigate the effect of varied substrate complement on removal efficiencies or hydraulic throughput. The smaller scale bench-top experiments identified increasing metal sorption to soil at increasing pH, with metal sorption found to be greatest in lead and lowest for zinc. These bench-top experiments are valuable in developing a knowledge base for rain garden substrate, but need to be substantiated in larger scale systems with monitoring of hydraulic parameters.

Field investigations have been performed periodically internationally; however with limited scope and duration. Davis et al (2007) investigated field-scale bioretention systems in College Park, Maryland with promising results. Median values for effluent event mean concentrations were 17 mg/L TSS, 0.18 mg/L total phosphorus, 0.004 mg/L copper, 0.004 mg/L lead, 0.053 mg/L zinc, and 0.02 mg-N/L of nitrate. Removal efficiencies were 47% for TSS, 76% for total phosphorus, 57% for copper, 83% for lead, 62% for zinc, and 83% for nitrate. The study was specifically designed to investigate the effects of an anoxic sump beneath the rain garden and thus effects of different substrates from the various systems were not investigated. Additionally, metal speciation (particulate and dissolved fractions) was not investigated making it difficult to assess toxicity of metal loads.

The Cooperative Research Centre for Catchment Hydrology at Monash University (Fletcher et al. 2004) reviewed a number of international studies and derived water quality expectations for infiltration and bioretention systems (Table 1-2). This report synthesised data from a wide range of treatment designs, but does not offer design recommendations. Additionally, the expected removal of heavy metals is 85%; however, the range is quite large (50-95%). A follow up to this report is necessary which identifies the most successful treatment designs and substrates to achieve optimum contaminant removal.

Table 1-2- Expected pollutant removal filtration systems (sand filters, biofiltration systems, infiltration systems) (Fletcher et al. 2004)

Pollutant	**Expected Removal (mean, range) (%)**	**Comments**
Litter and Organic Matter	100	Expected to trap all gross pollutants, except during high-flow bypass
TSS	85 (65-99)	Pre-treatment required to reduce clogging risk.
TN	64 (50-70)	Dependent on speciation and state (soluble or particulate)
TP	70 (40-80)	Dependent on speciation and state (soluble or particulate)
Coarse Sediment	(95-100)	May pose a clogging risk. These systems should have pre-treatment to remove coarse sediment prior to entry into the filter media.
Oil and Grease	N/A	Inadequate data to provide reliable estimate, but expected to be >75%.
Faecal Coliforms	N/A	Inadequate data.
Heavy Metals	85 (50-95)	Dependent on form (soluble or particulate).

In New Zealand, the ARC evaluated 41 rain gardens from 30 sites in the Auckland region (Jayaratne et al. 2010); however there is very limited data on contaminant removal coupled with hydraulic throughput. Few sites were monitored over a longer period of time for hydraulic conductivity, metals, nutrients, and TSS. (Jayaratne et al. 2010). The report is comprehensive on media depth (41 rain gardens), and hydraulic permeability (41 sites); however, only two sites were monitored with data for hydraulic conductivity, metals, nutrients, and TSS. These two sites (Waitakere City Vehicle Testing Station and Paul Matthews Rain Garden) are discussed herein.

The Waitakere City Vehicle Testing Station rain garden, one of the earliest rain gardens installed in New Zealand, was installed in 2000 by Landcare Research to manage runoff from the vehicle testing lanes and parking area. The rain garden is densely vegetated with flax plants and has a media depth of 300 mm, with less than 5% organic material in the substrate media (Jayaratne et al. 2010). The media is underlain by pumice, which has a very high hydraulic conductivity. The rain garden was monitored to investigate metal removal efficiency and hydraulic throughput. Researchers found high hydraulic conductivity (80-100 mm/hr) and good contaminant removal (83% TSS, 80% total Zn, 73% dissolved Zn, 51% for total Cu) over a 247 day period (Skeen et al. 2010). Removal efficiencies are high; however, no contaminant concentrations were provided to compare median discharge concentrations with relevant discharge guidance to evaluate performance. High contaminant removal does not necessarily mean the rain gardens were reducing contaminant concentrations down below allowable discharge values. Additionally, in larger storm events, a portion of the stormwater bypasses the rain garden completely and is discharged untreated. This untreated discharge is not accounted for in the reported rain garden removal efficiencies.

The Paul Matthews Rain garden in Auckland (installed in 2006) was designed with roughly a 40% reduction in footprint to the TP-10 guidance (which was considered an extremely conservative and limiting sizing guidance) (Smythe et al. 2007). The system was built with a 300-400 mm topsoil layer and an initial infiltration rate of 220 mm/hr. This subsequently dropped off to 120 mm/hr after one year likely due to high unmitigated sediment inflow. Removal efficiencies (96% TSS, 97% total zinc, 96% dissolved zinc, no reduction in total copper, leaching of dissolved copper) were promising (Jayaratne et al. 2010); however both

total and dissolved median metal contaminant concentrations discharged from the rain garden were above 90% ANZECC guidance. Total copper exceeded ANZECC guidance 11-fold. While the removal efficiencies are promising, additional research is needed on both rain garden substrate and design to achieve relevant discharge levels.

1.5 A Need for Research

International design manuals available for different rain garden applications (e.g. home garden and larger sub-catchment gardens) suggest using different proportions of sand and topsoil. It is believed that organic material can play a significant role in stormwater treatment in rain garden design although data on rain garden performance is very limiting (Davis et al. 2001; ARC 2003; Trowsdale and Simcock 2008; FAWB 2009; Carpenter and Hallam 2010). The only proposed design for rain gardens in New Zealand is given in the TP-10 document by the ARC (e.g. Figure 1-4), which is recognised as limiting and is undergoing revision. It is also unclear whether the design of rain gardens, a bioinfiltration system, falls under the 75% TSS removal objective, probably due to deficiencies in the current knowledge of the effectiveness of various devices (Smythe et al. 2007).

Table 1-3 summarises a comparison of rain garden design criteria currently available in the literature to highlight differences in design guidance. The table is not meant to be inclusive of all design manuals internationally, but does include influential and widely used design guidelines. Large differences are apparent from the limited guidelines (e.g. New Zealand systems propose a 13 mm/hr infiltration rate and >100cm topsoil (ARC 2003) compared with 13-130 mm/hr and >45 cm topsoil recommended in Californian systems (SFPUC 2009)) while these design recommendations do not seem to be informed by performance data.

Table 1-3- Comparison of international rain garden design guidelines

Design Guide	Infiltration Rate	Design Storm	Topsoil Depth	Sand/Gravel Depth
Auckland Regional Council (TP-10) (ARC 2003)	13 mm/hr minimum	1/3 of 2 year 24 hour rainfall	100 cm minimum	30 cm (optional)
Stormwater Source Control Design Guidelines, Vancouver, Canada (GVRD 2005)	6 mm/hr	3 months- 24 hour rainfall	45-120 cm	NA
Brisbane City Council Water Sensitive Urban Design (BCC 2007)	NA	3 months rainfall	30-100 cm	10 cm
Maryland Stormwater Design Manual (MDE 2009)	13 mm/hr minimum	NA	76-120 cm	15 cm
Facility for Advancing Water Biofiltration (FAWB 2009)	100 mm/hr minimum	NA	40-60 cm	15 cm
Seattle Public Utilities (SPU 2009)	150 mm/hr	NA	45 cm minimum	30 cm
Bioretention Performance, Design, Construction and Maintenance(NCSU 2006)	25-150 mm/hr	NA	90 cm	25 cm
San Francisco Stormwater Design Guidelines (SFPUC 2009)	13-130 mm/hr	NA	single layer: more than 45 cm	

While the use of rain gardens to treat stormwater runoff is increasing nationally, there is a surprising lack of knowledge about their water quality treatment and infiltration capacities (Lloyd et al. 2001; Fletcher et al. 2004; Dietz and Clausen 2005; Henderson et al. 2007; Trowsdale and Simcock 2008). Performance data for infiltration/ biofiltration systems from which to optimise their designs or predict future performance through model validation are still quite rare internationally (Fletcher et al. 2004). This provided a research opportunity to establish relationships between stormwater treatment performance and specific substrate type employed in rain garden bioinfiltration systems.

1.6 Research Objectives

The research objectives of this Master's thesis were to:

- Perform an in-depth literature review on design guidelines and stormwater management research including bio-infiltration system substrate optimization for both hydraulic and contaminant removal efficiencies.
- Investigate effects of topsoil proportions on (a) hydraulic conductivity and (b) metal removal efficiency in mesocosm-scale laboratory rain gardens using collected representative stormwater from a Christchurch catchment
- Investigate effects of pH supplementation provided by a waste product combined with organic material (i.e. topsoil) on (a) hydraulic conductivity and (b) metal removal in mesocosm scale laboratory rain gardens using collected representative stormwater from a Christchurch catchment.
- Assess metal speciation as a function of pH in effluent of the laboratory rain gardens.
- Monitor hydraulic throughput and metal removal efficacy in field-scale systems over a wet season (i.e. winter and spring).
- Contribute new knowledge to the efficacy of optimising bioinfiltration systems for stormwater treatment that could be useful to establish design standards in New Zealand.

1.7 Thesis Structure

The thesis structure includes an Introduction (Chapter 1), which includes background knowledge, context and research scope. The materials and methods section (Chapter 2) defines the approach, tools and quality assurance employed for the experiments to test the research hypotheses. Initially, mesocosm-scale rain gardens were established on which hydraulic and treatment experiments were conducted (Chapter 3). Based on results from these initial laboratory experiments, pH-augmented substrate experiments were designed and tested (Chapter 4). Field scale investigations of a 'rain garden' installed in Christchurch was conducted (Chapter 5) and considered in the discussion of results from the laboratory mesocosm-scale systems. It should be noted that the field system, while named a rain garden by the designer and owner, was not constructed in accordance with the design criteria that conventionally depict rain gardens (see Section 2.2.2). Finally, a summary of the research and future research recommendations are given (Chapter 6).

Chapter 2: Materials and Methods

2.1 Laboratory Mesocosm-Scale Rain Garden Experimental Design

Mesocosm-scale (180 L cylindrical, 0.17 m^2 surface area) rain gardens were established in a laboratory set-up (Figure 2-1). Water was pumped from the stormwater supply tank through peristaltic pumps to sprinkler diffusers. Water infiltrated through systems (shown in Figure 2-1 as 1-3) and 'treated' effluent was discharged through under-drain outlets. Substrate makeup was different in each of the three systems to investigate the effect(s) of organic topsoil on heavy metal (Zn, Cu, and Pb) and nutrient removal, as well as hydraulic throughput under simulated rain events. A small (20 mm) layer of bark mulch was applied on top of each system in order to help diffuse stormwater across the system as in full-scale rain garden construction practices. The volume of bark (upper surface) and under-drainage gravel (at exit) remained constant across the three systems. Sand (AP-20, well-graded coarse sand) and topsoil (sandy loam, 5% organic content) volumes were varied for all three systems (system 1: 500 mm sand; system 2: 500 mm topsoil; system 3: 250 mm of both sand and topsoil). Additional substrate characterization, including analytical data, is discussed in Section 2.1.3.1 and reported in Table 2-3. A total rain garden depth of 670 mm (with 520 mm of 'reactive' substrate) was maintained at the onset of the experiment.

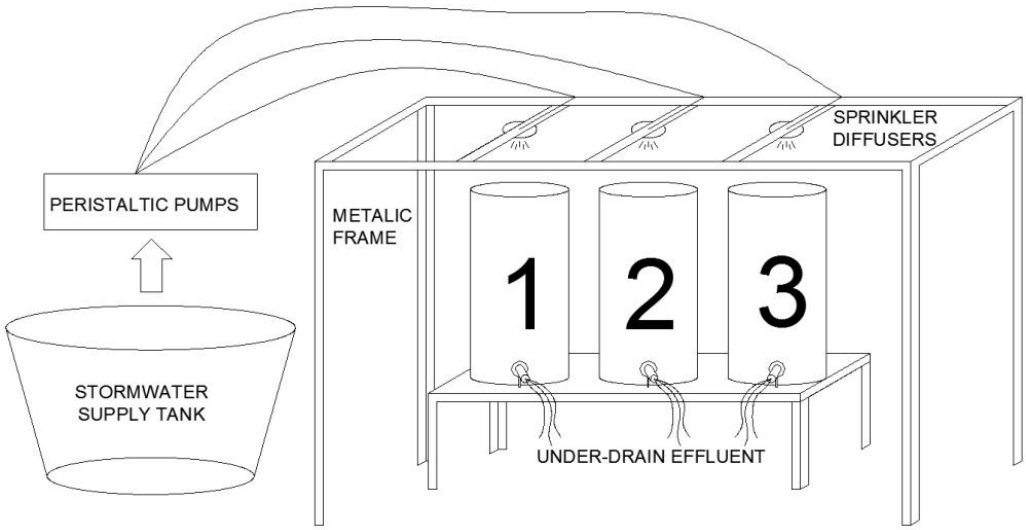

Figure 2-1- Laboratory-scale rain garden experimental setup.

2.1.1 System Decontamination

Mesocosm-scale rain gardens were established from three 180-L high-density polyethylene (HDPE) containers. All containers were initially decontaminated. The systems were scrubbed

with a stiff brush and warm water to remove large ferrous oxide deposits from previous use. The containers were then scrubbed with an Alconox solution, a phosphate-free concentrated, anionic detergent appropriate for cleaning HDPE and rinsed thoroughly with warm water. Containers were allowed to soak in a 5% orthophosphoric acid solution for 24-48 hours and then drained. This acid soak removed any residual metals and staining from the HDPE containers. A second Alconox solution was prepared and the containers were allowed to soak for approximately 5 hours. All containers were flushed a minimum of three times with tap water and then deionised water to complete the decontamination process.

2.1.2 System Construction

A stainless steel under-drain was installed at the side base of the decontaminated 180-L containers and fitted with a ball valve and a tap and hose connection. The valve created the potential to retain water in the system for extended periods of time (i.e. to test the effects of retention time and anoxic conditions on water quality treatment) and control hydraulic head. The hose connection also allowed the systems to be filled from the bottom, allowing entrapped air to escape upwards from the substrate and reduce the possibility for preferential flow paths (McCauley 2011).

The bases of the HDPE containers were filled up to the bottom of the under drain (70 mm-layer) with inert washed silica sand from Mt. Somers, Canterbury, New Zealand. The under drain outlet was unable to be installed at the very base due to constructability constraints (i.e. poor seal due to the curvature of the container), and thus the inert sand was installed to fill the space below the bottom of the under drain and prevent appreciable volumes of water from accumulating in the base. This inert layer was covered with poly-sheeting which was cut slightly oversized. Excess sheeting was neatly folded and forced against the sidewalls of the container to further prevent water from accumulating in the bottom of the systems rather than flowing directly out the under-drain. Above the poly-sheeting, 80 mm layer of rounded gravel (approximately 40 mm mean diameter) was placed above the top of the under drain to prevent clogging of the exit drainage point by providing an under-drainage zone. The upper level of the gravel layer was covered with Texpack-extra geotextile (sourced from Textile Bag & Sack Company Ltd in Sydenham, Christchurch), which served to prevent smaller particulates from the overlying topsoil from clogging the pore spaces of the gravel drainage layer. Although the geotextile rated permeability was 14 $L/m^2/s$, some geotextiles can limit hydraulic throughput.

To avoid such potential problems in the laboratory systems, the geotextile was perforated by cutting approximately ten 50 mm slit in the geotextile.

Substrate Composition

Above the geotextile, three different combinations of reactive substrate (bark mulch, topsoil and AP20 graded sand) were added according to Table 2-1 and Figure 2-2(A). The substrate material was placed in approximately 50 mm lifts and hand-compacted above which a 20 mm bark mulch layer was applied constituting the uppermost layer in each system. The total volume of gravel and bark mulch material in each system, and the initial total depth of the treatment system were constant. Diagrams showing the complete experimental setup are shown in Figure 2-1 and Figure 2-2.

Table 2-1- Laboratory-scale rain garden substrates

System Name	Depth of Reactive Substrate (mm)			
	Bark Mulch	Topsoil	Sand	Total System Depth[1] (mm)
Sand	20	0	500	670
Sand/Topsoil	20	250	250	670
Topsoil	20	500	0	670

Note: 1. Includes 150 mm depth gravel under drain layer.

Substrate pH-Supplementation

Additional laboratory systems later employed the same experimental set-up (520 mm of reactive substrate and total rain garden depth of 670 mm), but sand was replaced with varying amounts of coarsely crushed mussel shells in two of the systems as a pH amendment (the third system acted as a control treatment). Systems were constructed according to Table 2-2 and Figure 2-2(B).

Table 2-2- pH supplementation for laboratory-scale rain garden substrates

System Name	Depth of Reactive Substrate (mm)			
	Bark Mulch	Topsoil	Mussel Shells	Total System Depth[1] (mm)
TS	20	500	0	670
TS/MS (4:1)	20	400	100	670
TS/MS (1:1)	20	250	250	670

Note: 1. Includes 150 mm depth gravel under drain layer.

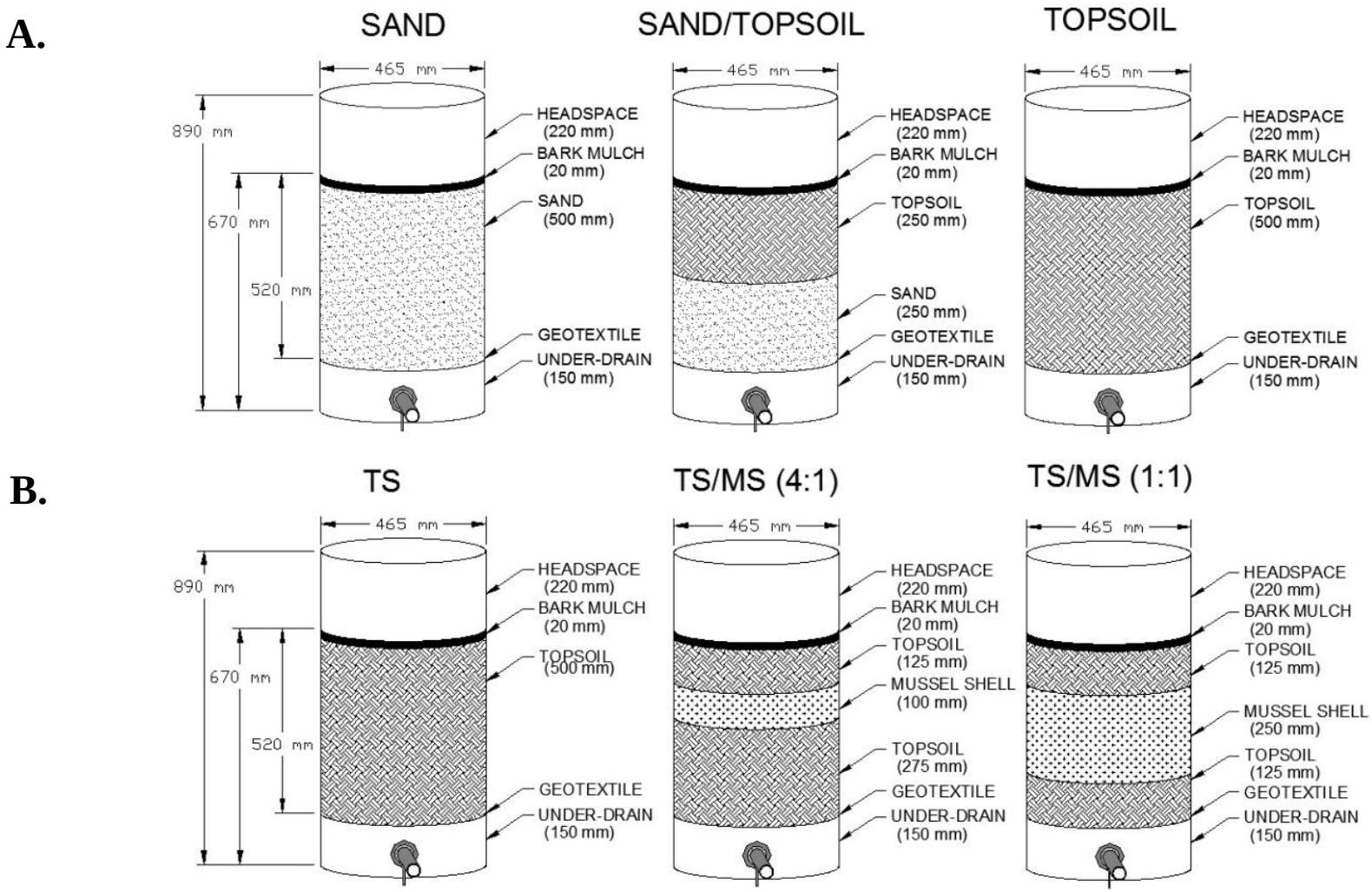

Figure 2-2- Mesocosm-scale laboratory rain garden construction for initial (A) and pH complement (B) experiments. Note: All systems were constructed with a total system depth of 670 mm of material of which 520 mm was active substrate.

2.1.3 Substrate Characterisation

AP20 Graded Sand

Graded unwashed sand was obtained from North End Sand and Shingle Supplies located in Woodend, Canterbury. Sand was graded to Christchurch City Council Stabilised AP-20 physical specifications as detailed in the sieve analysis (Figure 2-3). Sand grading was determined using the coefficient of uniformity (C_u) and coefficient of curvature (C_c) which are calculated using Eqn 2-1and Eqn 2-2, respectively.

$$C_u = \frac{D_{60}}{D_{10}} = 23.3 \qquad \textbf{2-1}$$

Where D_{60} = grain diameter at 60% passing (7 mm per Figure 2-3)

D_{10} = grain diameter at 10% passing (0.3 mm per Figure 2-3)

$$C_c = \frac{(D_{30})^2}{D_{10} \times D_{60}} = 1.9 \qquad \textbf{2-2}$$

Where D_{30}= grain diameter at 30% (2 mm per Figure 2-3)

Following the Unified Soil Classification System (ASTM), the AP20 sand is classified as a well graded sand (C_u>6 and 1<C_c<3). AP-20 sand is coarser than recommended filtration media specified in the FAWB (2008) design guidance; however it was used in Christchurch field rain garden systems and thus was employed in these laboratory experiments.

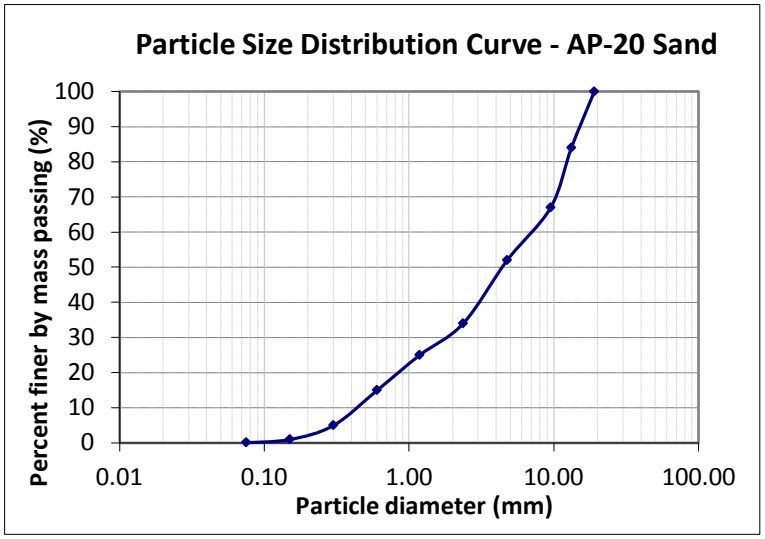

Figure 2-3- Particle size distribution curve for unwashed AP-20 sand from sieve analysis. Particle diameter (mm) is shown on a log-scale.

Organic Topsoil

Organic topsoil was obtained from Gardenmakers located in Sockburn, Christchurch. The topsoil was a standard screened top soil classified as sandy loam (USDA 1951) after sieve and sedimentation analysis performed in the UC Geotechnical Engineering Laboratory (Figure 2-4). Particle size analysis (PSA) for topsoil was confirmed using a Horiba laser scattering particle size distribution analyser LA-950 (see Appendix A). The D_{50} for the experimental sandy loam was 0.42 mm (Figure 2-4). Topsoil contained a 5% organic material (refer to Section 2.4.1.5 for analytical methodology).

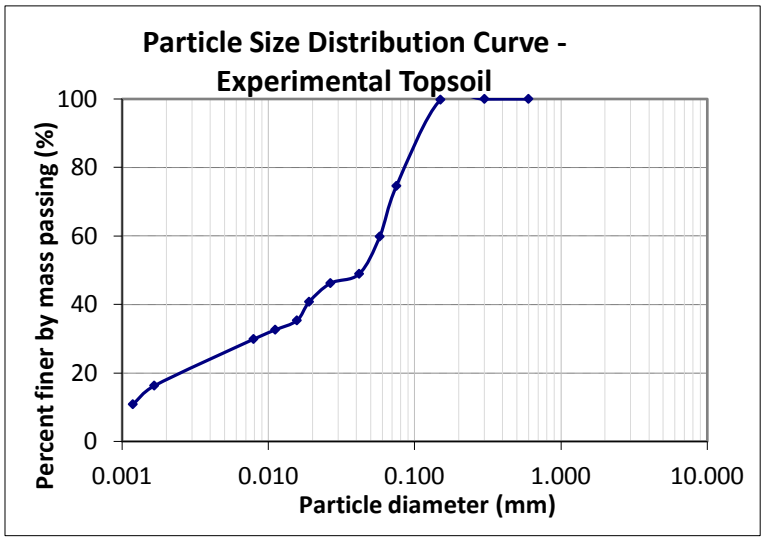

Figure 2-4- Particle size distribution curve for experimental organic topsoil from sieve analysis. Particle diameter (mm) is shown on a log-scale.

Bark Mulch

Bark mulch was obtained from Gardenmakers located in Sockburn, Christchurch. The mulch constituted an untreated mill bark mixture and served to distribute stormwater flow evenly across the surface reducing the potential for preferential pathways to occur vertically. While cobbles and course gravel can also be used to spread flow and reduce velocities, bark mulch is also capable of immobilising stormwater contaminants, which is not afforded by the more inert gravel.

Mussel shells

Mussel shells were only used in the pH-supplementation laboratory experiments as a source of alkalinity based on previous research that successfully employed the same mussel shells to mitigate acid mine drainage (McCauley et al. 2009). Mussel shells were sourced from United

Fisheries Ltd. in Sockburn, Christchurch and were sourced from both green lip and blue mussels. They were roughly crushed to approximately 30 mm mean fragments and were collected from the end of a mussel processing plant so residual mussel meat was attached to the shell as shown in Figure 2-5. This mussel meat was not removed from the shells for the experiments, as separation of mussel meat from shells was considered impractical and unnecessary in all scale applications.

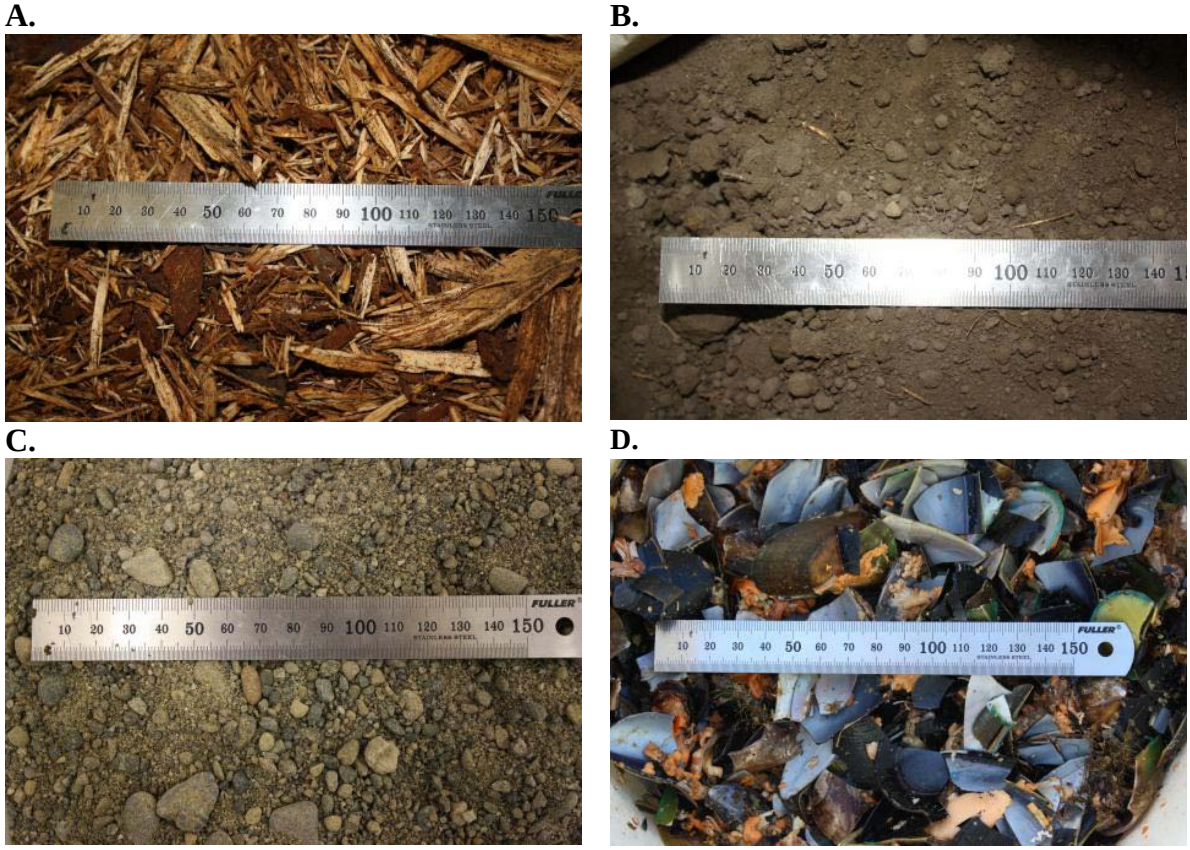

Figure 2-5- Substrates used in mesocosm-scale rain garden experiments including: A) bark mulch; B) sandy loam topsoil, C) AP-20 graded sand, and D) crushed mussel shells.

2.1.3.1 Substrate Sampling and Analysis

Prior to commencing the experiments, two composite fully homogenised samples per material type were collected into laboratory supplied certified clean containers and analysed to ensure substrates were not a source of the contaminants under investigation (Table 2-3). Baseline samples collected prior to experimental runs were analysed for total and toxicity characteristic leaching procedure (TCLP) Cu, Zn, and Pb, Total-N and cation exchange capacity (CEC). CEC was not performed for mussel shell and bark mulch material because the material would need to be ground prior to testing. Grinding up the bark mulch or mussel shell material would completely change the surface area of the material and thus the CEC data would be biased high. Analytical laboratory reports are included in Appendix B.

Table 2-3- Substrate baseline contaminant concentrations and leaching results. Mean values are shown compared with relevant guidance: ANZECC (2000) Interim Sediment Quality Guidelines (Low and High) and Resource Conservation and Recovery Act (RCRA 1976) TCLP toxicity limits.

Analysis	Relevant Guidance		Substrate			
			Bark Mulch	Topsoil	Sand	Mussel Shells
Total Recoverable Metals	**ANZECC ISQG**					
(mg/kg dry weight)	**Low**	**High**				
Cu	65	270	<4	6.8	5.0	<4
Zn	200	410	20.5	49.3	31.0	<8
Pb	50	220	6.3	15.5	6.8	<0.8
TCLP Metals (g/m3)	**RCRA TCLP Toxicity**					
Cu	25		<0.011	<0.011	<0.011	0.012
Zn	250*		0.065	<0.021	0.012	0.039
Pb	5*		0.003	0.004	<0.002	0.009
Total Nitrogen (g/100g dry weight)	-		0.336	0.217	<0.050	1.705
Cation Exchange Capacity (me/100g)	-		NA	11.5	3	NA

Notes: *= No RCRA toxicity guideline. Soluble threshold limit concentration values shown
<= Below detection limit
NA= Not applicable due to grinding procedure required to analyse for CEC.

Upon completion of the laboratory mesocosm-scale experiments, one composite sample was collected from the top 20 mm bark mulch layer from each of the three systems. Additionally, one composite sample was collected from each 25-45 mm substrate layer. This sample was from the sand layer in sand only system, and from the topsoil layer in sand/topsoil and topsoil only systems. All substrate samples were sent to Hill Laboratories for analysis. Post-experimental substrate sample results are reported in Table 3-10.

2.1.3.2 Substrate Leaching Investigations

Leaching experiments were setup in the laboratory as a result of nutrient export and pH differences observed in effluent midway through the treatment efficiency experiments. The purpose of these bench-top experiments was to assess (i) nutrient leaching and (ii) alkalinity potential from the substrates since it was not possible to continuously sample from each layer within the systems.

Equal volumes (300 mL) of substrates (topsoil, bark mulch, AP-20 grade sand) were added to 1-L beakers in a laboratory set-up (Figure 2-6). Filtered tap water (800 mL) was added to each beaker and the beaker was stirred for approximately 15 seconds. Experimental beakers were conducted in duplicate (shown in Figure 2-6) for quality control purposes. Discrete water samples were taken at five, ten and 30 minutes from the supernatant to analyse for nitrate and reactive phosphorus concentrations to assess leaching potential. Sample times (5-30 minutes) were comparable with estimated mesocosm residence time. Specific conductance and pH were monitored and recorded throughout the bench top experiments.

Figure 2-6- Bench top experimental setup for topsoil (left), bark mulch (centre) and AP-20 sand substrate (right).

Additional bench top experiments were established to quantify the effect of mussel shells on pH and to estimate the volumetric amount of mussel shells required to affect a pH change. Varying amounts of substrates were added to 12 different 1-L beakers each containing 700 mL stormwater (Table 2-4). The different treatments included uncrushed and crushed mussel shells and addition of topsoil. Supernatant pH readings were recorded using a calibrated EDT pH meter at 1, 5, 15, 30, 60 and 1080 minute intervals following mixing.

Table 2-4- pH amendment batch leaching experimental setup. Shading indicates the material was implemented in the beaker. Substrate volumes are shown in parenthesis next to each material.

Material	Beaker Number											
	1	2	3	4	5	6	7	8	9	10	11	12
Stormwater (700 mL)												
Soil (100mL)												
Soil (200 mL)												
Mussel Shells (100 mL)												
Mussel Shells Crushed (100 mL)												
Mussel Shells (200 mL)												
Mussel Shells Crushed (200 mL)												

2.1.4 Stormwater Used in Laboratory Experiments

2.1.4.1 Stormwater Collection

Experiments were conducted with actual stormwater rather than synthesised stormwater. Since field and lab systems are compared at the end of this thesis, it was optimal to use the same stormwater source to minimise introducing additional variables that could confound key parameters under investigation. Stormwater runoff for laboratory experiments was collected from the inlets of two installed field rain gardens (Section 2.2.1) in the Addington suburb of Christchurch during multiple storm events between 17 July 2010 (winter) and 1 September 2011 (spring). Approximately 160 L of water were collected on each sampling occasion. Representative storm water was collected across entire storm events including the first flush, first 10-30 minutes, as well as runoff later in the storm event. Influent storm water was collected into 20-L containers directly from the inlet pipe of the sump ensuring any suspended solids/sediment in the stormwater were also collected into the sample containers. All water was transported back to the University and stored at <4°C prior to experimental runs. Event mean concentration (EMC), a measure of flow-weighted contaminant concentrations over a storm event and typically applied to stormwater monitoring calculations (Davis et al. 2001;

Lee et al. 2004; Herngren et al. 2005) were not possible due to constraints involved with monitoring stormwater flow into the rain garden (further discussed in Section 5.1.2).

2.1.4.2 Water Quality Volume

Laboratory mesocosm-scale rain gardens were sized (as per Eqn 2-3) according to ARC TP-10 (2003), which are the most relevant current design guidelines for rain gardens in New Zealand.

$$A_f = \frac{WQV * d_f}{K * (h + d_f) * t_f} \quad \textbf{2-3}$$

Where A_f = surface area (m2)

WQV = treatment volume (m3)

d_f = planting soil depth (m)

K = coefficient of permeability (m/day)

h = average height of water (m)

t_f = time to pass WQV (days)

The Water Quality Volume (WQV) was calculated from runoff calculated in accordance with ARC TP-108 (ARC 1999). The runoff generated from the design storm specified in TP-10 (1/3 of 2 year, 24 hour annual recurrence interval event = 15.33 mm in Christchurch (NIWA 2010)) was calculated using Eqn 2-4 from TP-108 based on the Soil Conservation Service (SCS) rainfall-runoff curves used to describe rainfall losses (SCS 1986).

$$Q = \frac{(P - Ia)^2}{(P - Ia) + S} \quad \textbf{2-4}$$

Where Q= runoff depth (mm)

P= rainfall depth (mm)

S= potential maximum retention after runoff begins (mm)

Ia= initial abstraction (mm)

Unlike the SCS guidelines (SCS 1986) which suggest an initial abstraction related to the soil storage parameter (Ia = 0.2S) based on empirical data, TP-108 recommends a constant initial

abstraction relative to the impervious area, with Ia =5 for pervious areas and Ia = 0 for impervious areas. Using 75% impervious area (calculated from aerial images of Addington field rain garden catchment), Ia = 1.25 mm.

S was calculated using the SCS curve number, CN, which is related to soil and land-use conditions of the catchment. The SCS curve number was determined using a weighted average of impermeable area (CN=98) and lawn with poorly draining soils representative of Christchurch (CN = 79) and assuming average antecedent moisture conditions (Mays 2004). A curve number of 94 was used for the catchment draining to Addington field rain garden system (75% impermeable area). While the curve number was relatively high, it was representative of the Addington catchment which has a low initial abstraction (potential maximum retention, S). The potential maximum retention after runoff begins, S, was calculated from curve number, CN, using Eqn 2-5 from TP-108.

$$S = \left(\frac{1000}{CN} - 10\right)25.4 = \left(\frac{1000}{94} - 10\right)25.4 = 16.21 \text{ mm} \qquad \textbf{2-5}$$

Substituting P (15.33 mm), Ia (1.25 mm) and S (16.21 mm), into Eqn 2-3, the runoff depth, Q, was calculated to be 6.54 mm. The WQV was calculated assuming an area of 5,300 m^2 (catchment size draining into a representative Addington field rain garden) and multiplying it by the runoff depth (6.54 mm). The WQV was calculated to be 34.66 m^3.

The surface area of the design rain garden, A_f, in Eqn 2-3, was calculated using the WQV above (34.66 m^3) and recommendations from TP-10 for d_f (1 m), K (0.3 m/day), h (0.11 m), and t_f (1.5 days). A_f was calculated to be 69.39 m^2.

Since the pre-defined cross sectional area of laboratory systems was 0.17 m^2, a scaling down factor was required to relate calculated field size following the ARC TP10 design guidance to the restricted lab size systems (Eqn 2-6).

$$\text{Scaling factor} = \frac{\text{mesocosm}}{\text{field}} = \frac{0.17\text{m}^2}{69.39\text{m}^2} = 0.00245 \qquad \textbf{2-6}$$

The required laboratory WQV was computed assuming an 8 mm rain event (equal to or larger than 85% of rain events in Christchurch (NIWA 2010)). The WQV was calculated by first determining Q using Eqn 2-4 (P = 8 mm, Ia = 1.25 mm, S = 16.21) and multiplying by the field catchment area (5,300 m^2) and the scaling factor (0.00245). The laboratory WQV was 25.8 L.

2.1.4.3 Stormwater Application

Stormwater collected in the field was transferred into a 350-L feed container and mixed with a large stirrer to ensure homogeneity. The stormwater was allowed to settle for approximately 5 minutes prior to each experimental run to simulate sedimentation sump conditions (replicating standard field system operation) designed to allow the heavier particulate matter to settle out of the stormwater and prevent clogging of the treatment system.

During the experimental runs, stormwater was fed to each system via a peristaltic pump through a Spectrum 360º Adjustable Flow sprinkler diffuser, which spread the stormwater evenly over the mesocosm scale rain gardens. The WQV calculated in Section 2.1.4.2 was applied over a 73-minute time period (48-min. storm event plus 25-min. calculated time of concentration). The 48-min storm period was derived using a simulated 8 mm rain event at a steady rainfall intensity of 10 mm/hr. An 8 mm rain event is greater than 85% of rain events in Christchurch between 2005 and 2010 and 97% of annual rainfall in Christchurch is less than 10 mm/hr (NIWA 2010).

Time of concentration (Eqn 2-7) was estimated using the Federal Aviation Administration method (Mays 2004). This method is commonly applied to overland flow in urban basins (Mays 2004).

$$T_c = \frac{1.8(1.1 - C)L^{0.50}}{S^{0.33}} = 25 \text{ mins} \qquad \textbf{2-7}$$

Where T_c= Time of Concentration (mins)

C= rational method runoff coefficient (0.8) (Mays 2004),

L= length of overland flow (ft) (<250 ft),

S=surface slope (ft/ft) (4%).

Applying 25.8 L of water over 73 minutes equates to an average flow rate of 0.35 L/min. Sprinklers and peristaltic pumps were calibrated in the lab to convey 0.35 L/min of stormwater (Figure 2-7).

Figure 2-7- Calibration of peristaltic pump and sprinkler diffuser.

2.1.5 System Flushing

Rain gardens, like most systems containing organic material, can net export carbon and nutrients when first operating as newly saturated biosystems. Therefore, all laboratory systems were initially flushed 3-5 times with tap water filtered through a 10 µm inline cartridge prior to the experiments. During flushing, effluent samples were periodically taken and monitored for pH, turbidity, nitrate, and specific conductance. Flushing was stopped once these parameters had stabilized in the effluent, usually after 2-3 hours. Up to 30% compaction (determined by depth difference) was observed in each system containing topsoil due to settling but less (3-10%) settling was observed in sand-only systems.

2.1.6 Constant-Head Saturated Hydraulic Conductivity Experiments

After flushing, each system was then saturated by filling with the filtered tap water from the bottom to allow entrapped air to escape upwards, reducing the possibility for preferential flow paths to occur. If air was hypothetically trapped in the substrate, hydraulic conductivity results would be bias low (Das 2009); however, slowly filling the systems from the bottom reduced the risk of preferential paths. The inflow rate and flexible stand pipe were adjusted to maintain a constant head of 110 mm over the substrate (Figure 2-8).

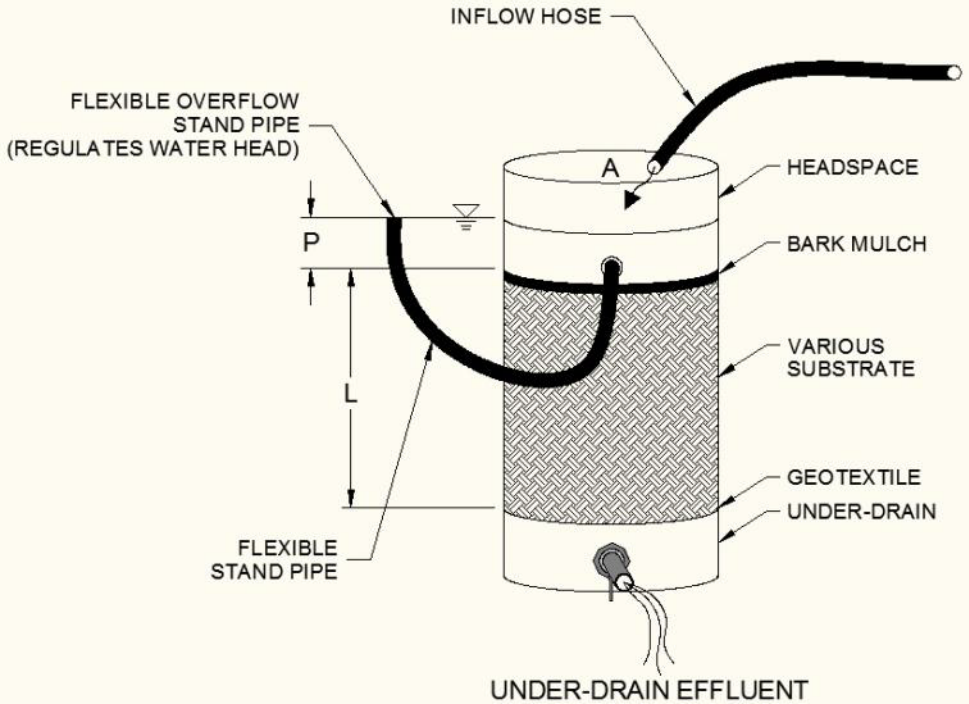

Figure 2-8- Constant head hydraulic conductivity experimental setup for mesocosm-scale laboratory systems. P is the water depth overlying the substrate, L is the substrate depth, and A is the cross sectional area.

Downward flow rates through the saturated systems were then determined using a stopwatch and a 100 mL graduated cylinder (n=25 measurements per system). A derivation of Darcy's equation (Eqn 2-8) was used to calculate the saturated hydraulic conductivity for each system.

$$K_{sat} = \frac{Q \times L}{A \times (L + P)} \qquad \textbf{2-8}$$

Where: K_{sat} = saturated hydraulic conductivity (m/s),
Q = flow through the saturated substrate (m^3/s)
L = depth of the substrate layers (m)
A = cross sectional area of the substrate (m^2)
P = water depth overlying the substrate (m)

Saturated hydraulic conductivity was also tested periodically at two additional constant head levels (75 and 130 mm) to ensure there were no changes in K_{sat} as a result of changing water head. Hydraulic conductivity tests were later repeated in the same manner following the completion of the water quality treatment experiments (Section 2.1.7) to ascertain potential differences in hydraulic conductivity after water quality 'treatment' occurred when multiple stormwater volumes were employed.

2.1.7 Water Quality Treatment Experiments

Untreated stormwater runoff from a neighbouring Christchurch city catchment (where an operational rain garden was concurrently monitored) was collected (Section 2.1.4.1) and applied to the mesocosm scale laboratory systems (Section 2.1.2). The calculated WQV (Section 2.1.4.2) was applied over a 73-minute time period (Section 2.1.4.3). Initially, a conservative 'standard' contaminant loading rate (comprising metal concentrations of Cu: 5.99 ± 0.73 µg/min, Zn: 57.89 ± 6.06 µg/min and Pb: 13.65 ± 2.80 µg/min) was applied. This application rate was duplicated in a second experimental run, after which it was followed by a 'higher' (i.e. doubled contaminant concentrations) loading rate to ascertain whether the systems had reached their maximum removal capacity, which would be evident from break-through in treatment behaviour.

In the mesocosm-scale alkalinity supplementation experiments, a new batch of stormwater collected from storm events on 8 July 2011 and 1 September 2011 was applied in triplicate. The 'standard' contaminant loading rate was Cu = 8.88 µg/min, Zn= 35.59 µg/min, Pb = 11.39 µg/min.

2.2 Field-scale Catchment Description and Experimental Design

2.2.1 Catchment Characteristics

All stormwater experiments performed during this research employed representative stormwater collected from the Addington catchment, an urban neighbourhood of Christchurch, New Zealand (Figure 2-9) where field systems were concurrently monitored. The two sub-catchments (#12 and 13) delineated in Figure 2-9D are approximately 2,500 m^2 each. Rain gardens #12 and 13 are located at the corner of Selwyn Street and Fairfield Avenue, 1.8 kilometres southeast of the Christchurch city centre (43°32'39.86"S, 172°37'22.58"E). The area is predominantly residential, within an urban catchment with approximately 75% impervious area (roads, roofs, driveways, paths, etc.).

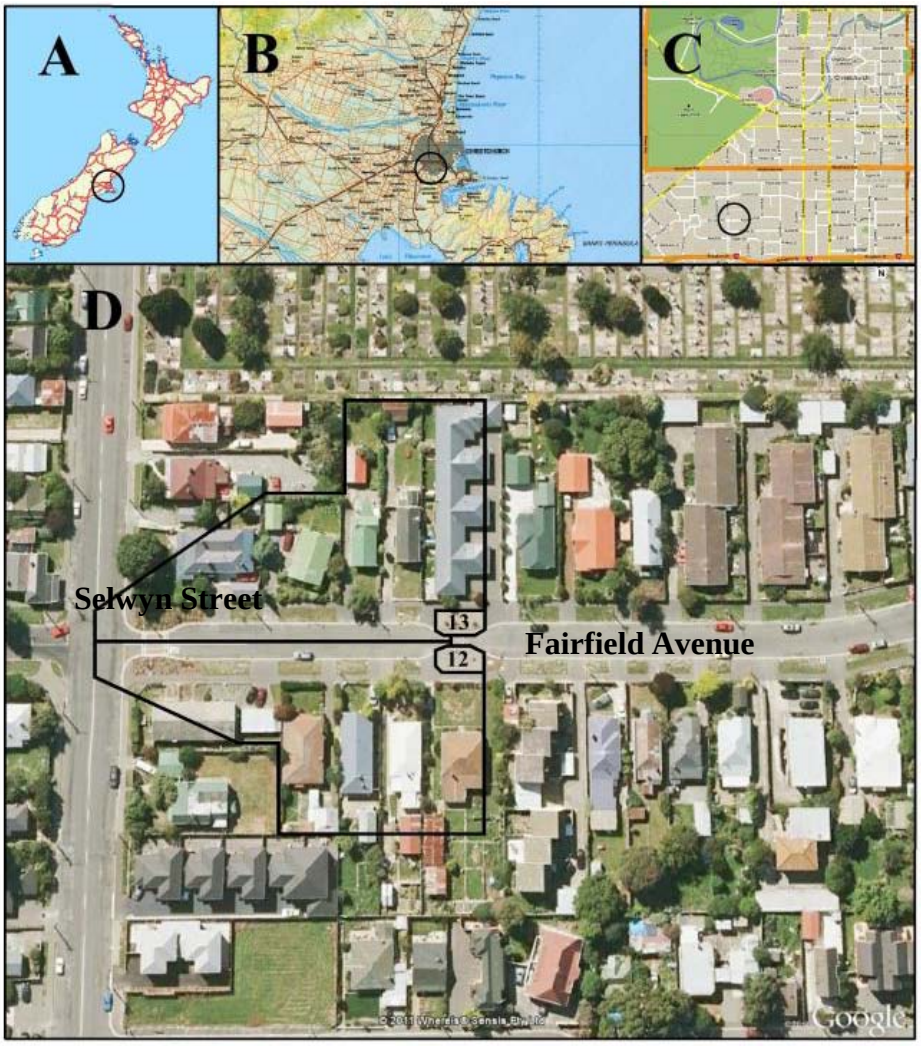

Figure 2-9- Site location map including: A) New Zealand, B) Christchurch City, C) Addington Suburb, D) rain garden sub-catchments 12 and 13 .

2.2.2 Rain Garden Design Specifications

Nineteen "rain gardens" were installed in Addington, Christchurch in 2006-2007 as part of a stormwater modernisation programme of some of Christchurch's oldest suburbs. The installed systems were not sized according to ARC TP-10 sizing recommendations due to space constraints (shallow buried utilities, parking demand, high water table, shallow sloping topography), which may also explain their deviation in design from conventional rain garden designs reported elsewhere. Rain gardens #12 and 13 (Figure 2-9) were targeted for stormwater collection used in laboratory experiments (Section 2.1). Additionally rain garden #12 was the focus of a more detailed field-scale investigation, which included flow monitoring (Section 2.2.3) and analytical sampling (Section 2.3.1). Figure 2-10 shows a schematic of a representative field scale system installed in Addington. Water flow through

the system is indicated by arrows and generally flows from left to right. Detailed as-built drawings are included in Appendix C. The reason these systems were constructed without topsoil was because it was believed that the reduced hydraulic conductivity of topsoil would require a larger areal footprint, which the development could not offer, to treat the same volume of stormwater in the targeted catchment. While the field-scale rain gardens are not a standard rain garden configuration, there is still treatment through the AP20 graded sand located downstream of the percolation chamber (hollow HDPE tank with permeable, perforated walls wrapped in geotextile fabric) designed to detain water to reduce peak flows and total volume.

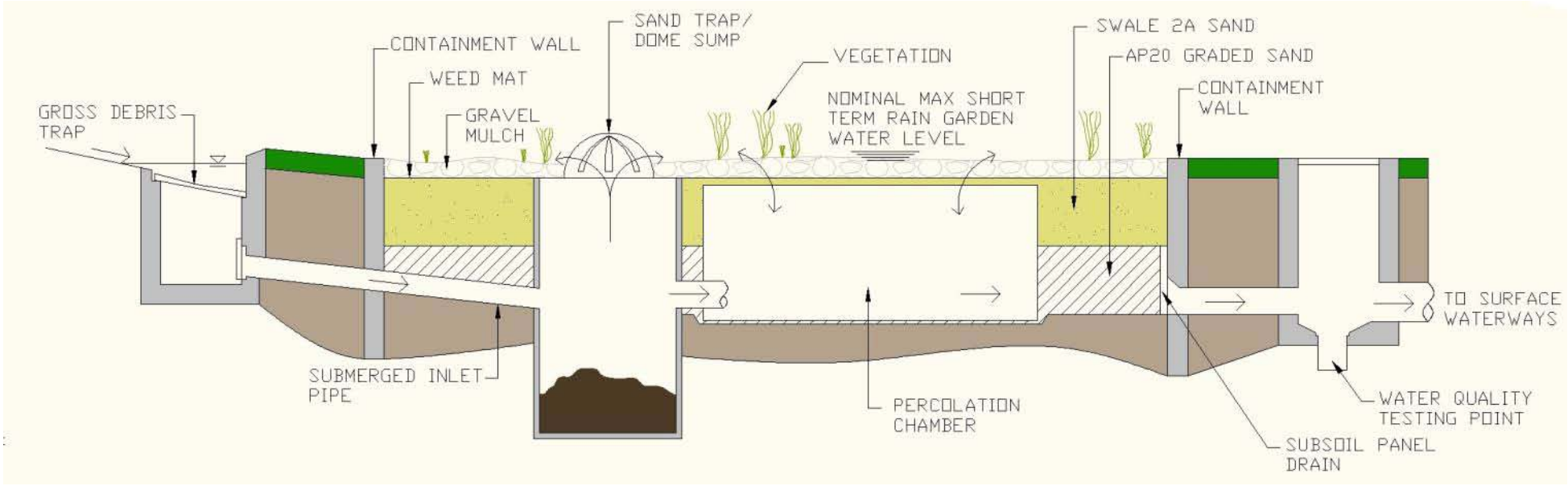

Figure 2-10- Field-scale 'rain garden' schematic (modified from as-built drawings provided by CCC). Stormwater flow is indicated by arrows and generally flows from left to right. The 'gross debris trap' and 'sand trap' serve as stormwater pre-treatment prior to sand filtration.

2.2.3 Flow Monitoring Set-up

Field rain garden #12 was fitted with Odyssey capacitance water level probes at the inlet sump (Sand Trap/Dome Sump in Figure 2-10) and the outlet sump (Water Quality Testing Point in Figure 2-10) to monitor water flow through the system (Figure 2-11). Inlet and outlet weirs were also constructed to increase the accuracy of flow rate measurements. The inlet sump contained two 150 mm polyvinyl chloride (PVC) pipes and two similar outlet pipes, which conveyed water to the rain garden. One of the outlet pipes was blocked off with a bung while the second pipe was fitted with a 97.5 degree weir for more accurately measuring the inflow.

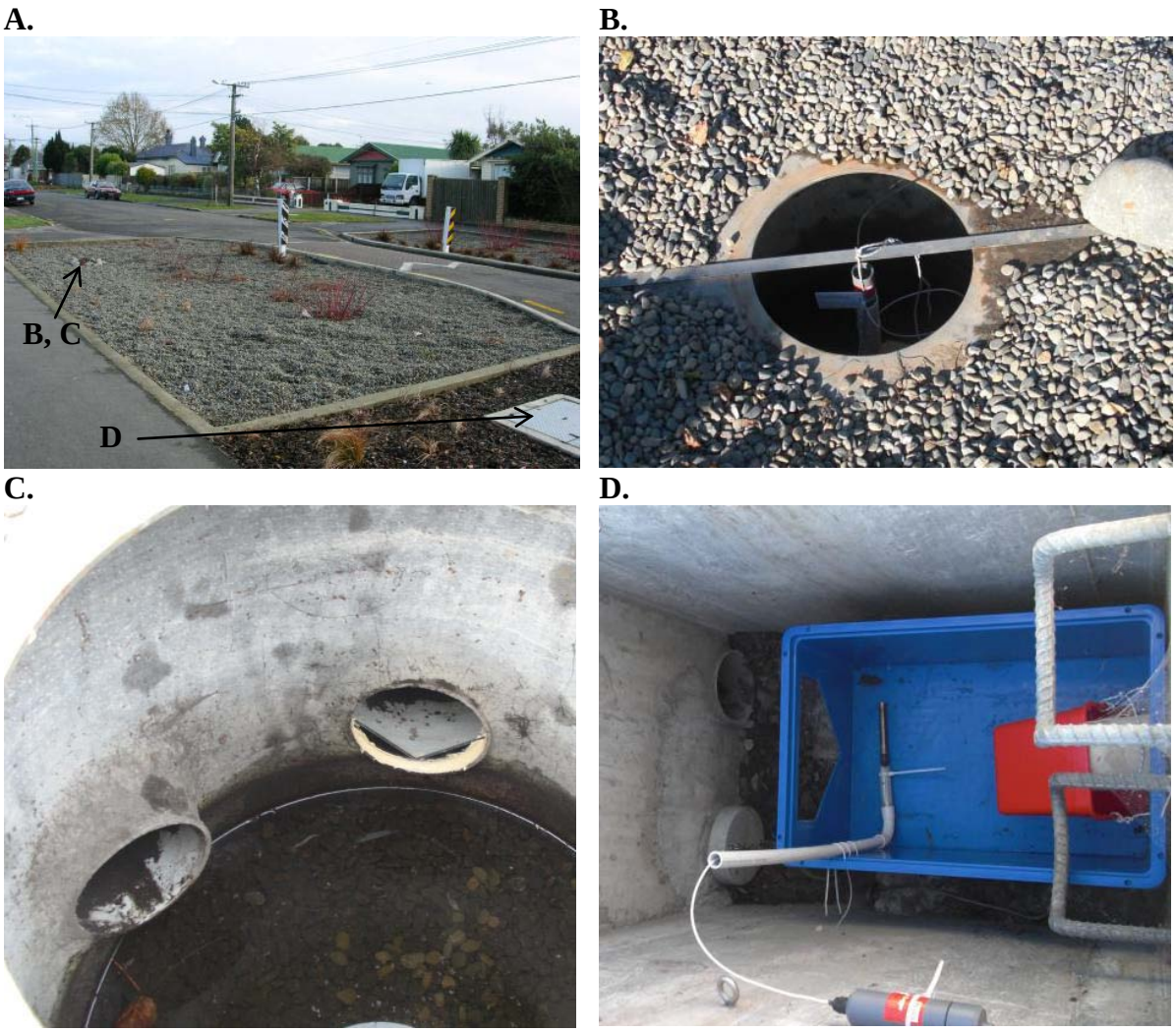

Figure 2-11- Operational field-scale rain garden with flow gauging at inlet and outlet. Water level data loggers were installed at the inlet (B, C) and outlet (D) structure of Addington rain garden #12 (A) which recorded depth over 97.5 degree weirs. Location of pictures B, C, and D are shown in A.

2.2.3.1 Flow calibration

Prior to installation in the field, water level probes were calibrated in the UC Environmental Engineering Lab using the tracer mode on the water level probes and a bucket. The water level probe was marked at 10 and 100 mm increments. The probe was placed in a bucket of water up to the two marks and capacitance readings were recorded at the two different levels. A linear calibration file (capacitance-depth relationship) was saved in the Odyssey programme and applied to the baseline capacitance readings to obtain equivalent water levels.

All weirs were calibrated in the laboratory using a magnum flow meter and a point gauge. The water level height and discharge volume were recorded to develop stage discharge relationships for each weir. Results of calibration are shown in Appendix D. Stage discharge relationships were compared with theoretical values calculated using Eqn 2-9.

$$Q = \frac{8}{15} C_d \sqrt{2g} \tan\frac{\theta}{2} H^{\frac{5}{2}} \quad \textbf{2-9}$$

Where Q= volumetric flow rate (m^3/s)

C_d=Coefficient of discharge

g= acceleration due to gravity (9.81 m/s)

θ= angle of v-notch weir (degrees)

H= vertical depth at the v-notch (m)

For a 97.5 degree weir, a C_d of 0.58 was assumed (Cd ranges from 0.57 to 0.59 for fully contacted weirs with a notch angle between 20 and 100 degrees). Consequently, the equation simplified to Eqn 2-10.

$$Q = \frac{g^{0.5}}{2} \times H^{\frac{5}{2}} \quad \textbf{2-10}$$

Inlet and outlet water level probes were installed in the field rain garden and offsets were obtained to ensure a zero reading was recorded at the base of each v-notch weir. This was accomplished by placing a sharp pointed cone at the far side of the sump upstream from the weir and adjusted to sit exactly level with the base of the v-notch. Water was added to the sump until the cone was creating the slightest dimple in the surface of the water. A water

capacitance reading was then taken and recorded as the offset to the bottom of the weir (if water was added until it was discharging over the weir, the offset would be incorrect due to surface tension effects impacting water discharging over the weir).

Although the rain garden inlet was equipped with a water level data logger and v-notch weir, exact flow estimates were not possible due to constrained discharge through the percolation chamber likely due to the geotextile surrounding the percolation chamber (pers. comm. with system designer (Wehrmann 2011)) (refer to Figure 2-10). Stormwater discharge was restricted leaving the percolation chamber, and thus backed up into the inlet sump. The stage discharge rating equation (Eqn 2-10) assumes a free flow downstream of the weir, and thus does not apply for the inlet of the rain garden. Runoff volumes entering the rain garden are thus estimates based on catchment characteristics.

Runoff volume was estimated in accordance with ARC TP-108 (1999) using the SCS curve number which is the most appropriate and straightforward method to estimate flow volumes entering the rain garden. Runoff calculations are consistent with previous runoff calculations presented in Section 2.1.4.2. Runoff was calculated in accordance with Equations 2-4 and 2-5. SCS curve number was determined using a weighted average of impermeable area (CN=98) and law with poorly draining soils representative of Christchurch (CN = 79) and assuming average antecedent moisture conditions (Mays 2004). A curve number of 94 was used for the catchment draining to rain garden #12 (75% impermeable area), consistent with previous runoff calculations (Section 2.1.4.2).

2.2.4 Precipitation data

The primary source for precipitation data was the HydroEco weather station located outside the UC Environmental Engineering Laboratory. The station is approximately 4 km northwest of the rain gardens so minor variation in rain patterns may exist between the field systems and weather station due to this distance. Although there is a NIWA weather station closer to the Addington rain gardens station located in the Botanic Gardens immediately adjacent to Hagley Park (about 1.6 km from the field systems), the UC weather station was preferred due to its high resolution. Total daily precipitation data were compared for the two stations upon completion of the research and there was less than 5% difference in the total daily precipitation values.

UC weather station precipitation data were recorded using an OTT (Germany) Parsivel laser-optic disdrometer capable of detecting rainfall intensity down to 0.001 mm/hr. The scanner is set to record intensity every minute. Due to intermittent power cuts (due to the series of major earthquakes in 2010-2011 and routine maintenance) throughout the course of this research, the laser scanner was periodically unavailable. When the scanner was not operating, precipitation data from the NIWA weather station were utilized.

2.3 Sampling Methodology

2.3.1 Sample Collection and Quality Assurance/Quality Control

Water was manually sampled following the contextual Australian and New Zealand Environment and Conservation Council (ANZECC) guidelines (ANZECC 2000). In compliance with these guidelines, at least 10% of the samples were duplicated for Quality Assurance/Quality Control (QA/QC) purposes. Standard, blank, and spike samples were carried out regularly. All equipment was calibrated in accordance with manufacturer's guidelines. Samples were collected head-space free in high-density polyethylene (HDPE) sampling bottles. All samples were placed into chilly bins or refrigerators and stored at <4°C prior to analysis. All samples were analysed within applicable holding time requirements except where otherwise noted. Water quality parameters including pH, specific conductance and turbidity were measured in the field using portable water quality meters (See Section 2.3.2) and recorded in a field book.

Laboratory Mesocosm-scale System Sampling

Influent stormwater samples were collected from the feed tank prior to commencement of each experimental run through the mesocosm scale rain garden experiments. Effluent samples were collected every five minutes for the first 30 minutes and thereafter every 20 minutes until the systems were drained to field capacity (i.e. all excess water had been drained and the systems were no longer dripping).

Field Sampling

Field inlet sump samples were collected from rain garden #12 directly from the inlet pipe (Figure 2-10) to ensure any suspended material was captured. Samples were collected every

ten minutes from the inlet pipe across each sampling storm event. Water exiting the rain gardens was collected in an outlet sump and these samples were collected periodically across the storm event. A total of three outlet samples were collected from each storm event due to low stormwater volume discharged from the rain garden outlet.

2.3.2 Calibration and Operation of Portable Water Quality Instruments

Water quality parameters (including temperature, pH, specific conductance, turbidity and nitrate) were measured using calibrated portable water quality instruments. A calibrated logging YSI multimeter was also placed in the feed tank and programmed to log every second for a minimum of five minutes for specific conductance, nitrate, pH and temperature. Analytical instruments were properly calibrated and maintained in accordance with manufacturer's guidelines. Instruments were calibrated with fresh standards prior to experimental runs and verified after sampling to ensure accuracy of measurements throughout the sampling.

The pH probe was calibrated daily with fresh 4.0, 7.0 and 10.0 SU buffers. Due to demand for the multimeter by other researchers, pH was periodically measured using EDT pH meters (Series 3) with their probes also calibrated with fresh 4.0, 7.0 and 10.0 SU buffers. There was no discernable difference between measurements from the YSI and EDT pH meters. Calibration for nitrate measurements was conducted daily using fresh 1 and 100 mg/L NO_3 solutions and checked periodically against spectrophotometer values to ensure accuracy. Turbidity was measured using a Hach Model 2100P portable turbidimeter. Calibration was performed with Hach StablCal® calibration standards or freshly prepared formazin standards. Specific conductance calibration was verified against a standard of 0.01 M KCl (1412 µS/cm @ 25 degrees C). Specific Conductance was internally calculated within the probe from electrical conductivity normalised to 25°C using Eqn 2-11.

$$\text{Specific Conductance } (25°\text{C}) = \frac{\text{Conductivity}}{(1 + \text{TC} * (\text{T} - 25))} \qquad \textbf{2-11}$$

Where TC= constant (0.0191),

T= temperature (°C)

2.4 Analytical Methods

2.4.1 Substrate Analysis

Previous studies identified metal contamination in compost (mainly arsenic (Rutherford et al. 2003)) and bark (mainly copper (Townsend et al. 2003)) mulch materials, resulting in net export of these metals from treatment systems in which those substrates were employed. Therefore, substrates were analysed for metals before the experimental systems were established in the laboratory to ensure they were not a potential source of metals. These data also provided a baseline value for each substrate type prior to commencing treatment experiments. Additionally, select substrate samples were later analysed following the first mesocosm laboratory experiments to ascertain the level of metal accumulation within the substrates (see Section 3.1.6 for further details).

2.4.1.1 Total Recoverable Metals

Total recoverable copper, zinc, and lead from each substrate type and from the post-experiments substrate mixes were analysed by USEPA Method 200.2 (USEPA 1994). Analysis of the digest was performed using ICP MS according to SM3125 (APHA 2005).

2.4.1.2 Toxicity Characteristic Leaching Procedure

Toxicity characteristic leaching procedure (TCLP) (USEPA Method 1311) is a material testing procedure to determine the mobility of both organic and inorganic analytes present in liquid, solid and multiphasic substrates (USEPA 1992). The procedure originated as a waste characterisation test to determine if a waste meets the definition of a toxic material under the United States Resource Conservation and Recovery Act (RCRA 1976). TCLP testing was performed on topsoil and bark mulch substrates by RJ Hill Laboratories, an International Accreditation New Zealand (IANZ) laboratory to ascertain the potential for contaminant leaching from individual and mixed substrates. Substrate TCLP results were presented in Table 2-3.

2.4.1.3 Cation Exchange Capacity

Since adsorption plays a major role in biofiltration systems, it was important to measure the cation exchange capacity (CEC) of the materials used in the mesocosm-scale experiments. CEC is determined by, and is a function of, the number of adsorption sites. The CEC of the

bark mulch and mussel shells was not tested because the test would completely change the surface area of the material. Substrate CEC results were presented in Table 2-3.

2.4.1.4 Total Nitrogen

Nitrogen was included in this research because of the potential for conversion to nitrate-N through mineralization and nitrification (Randall and Mulla 2001). Nitrate-N has been found to leach to surface waters through subsurface drainage (Randall and Mulla 2001) similar to rain garden under-drains. Total nitrogen (Total-N) was analysed according to SM4500 (APHA 2005) and is calculated using Eqn. 2-12.

$$\text{Total(N)} = \text{Nitrate(N)} + \text{Kjedhal Nitrogen} \qquad \textbf{2-12}$$

2.4.1.5 Organic Matter

Organic matter was roughly approximated from fixed and volatile solids analysed by SM2540 (APHA 2005). Topsoil samples were first oven dried at 180°C to evaporate any residual moisture and then ignited at 550°C to volatize any organic matter. Organic matter is estimated as the difference in weight between the ignited and oven dried samples.

2.4.2 Water Quality Analysis

2.4.2.1 Ionic Activity: pH and Specific Conductance

Stormwater pH, specific conductance, water temperature and nitrate were recorded using an YSI professional plus water quality meter outlined earlier in Section 2.3.2.

2.4.2.2 Metals: Total and Dissolved (Copper, Zinc, Lead)

Total metal samples were preserved with concentrated nitric acid (HNO_3, Fisher, trace analysis grade) to reduce the pH to less than 2.0 (APHA 2005). All metals (total and dissolved Cu, Zn, Pb) were analysed by ICP-MS (Agilent) following SM 3125B (APHA 2005). Total metal samples for digestion were mixed thoroughly on a magnetic stir plate while 25 mL of sample were transferred to a 50 mL polypropylene centrifuge tube. After the addition of 5 mL concentrated HNO_3, tubes were placed in a heating block and samples were boiled for one hour. Cooled samples were then filtered through an encapsulated 0.45 µm PVDF filter (47 mm, Labserv) directly into the analysis tube and analysed via ICP-MS. Dissolved metal samples were pre-filtered through disposable 0.45 µm filters before the HNO_3 acidification.

2.4.2.3 Nutrients: Nitrate and Reactive Phosphorus

Nitrate and reactive phosphorus were analysed on a Hach DR2000 or Hach Odyssey spectrophotometer using standard reagents. Nitrate-nitrogen (NO_3-N) was analysed in accordance with Hach (2003), based on the cadmium reduction method. Reactive phosphorus was analysed on Hach spectrophotometer using Molybdovanadate reagent in accordance with Hach (2003). Reactive phosphorus analysis was discontinued due to low levels in stormwater and leaching experiments.

2.4.2.4 Hardness

Although stormwater typically has low hardness, it was important to confirm because as hardness increases, ANZECC trigger values increase and ecotoxicity decreases accordingly (Section 1.3). Water hardness was analyzed according to SM2340 (APHA 2005) using an EDTA Titrimetric method. Hardness is defined in Eqn 2-13.

$$\text{Hardness, [mg equivalent}\frac{\text{CaCO}_3}{\text{L}}] = 2.497\left[\text{Ca}, \frac{\text{mg}}{\text{L}}\right] + 4.118\left[\text{Mg}, \frac{\text{mg}}{\text{L}}\right] \quad \textbf{2-13}$$

EDTA was titrated into a mix of 25 mL sample, 25 mL distilled water, 2 mL Ammonia buffer, and 1-2 drops indicator solution (comprising Eriochrome Black T dissolved in triethanolamine). If the solution contained calcium and magnesium ions at a pH of 10, the solution turned wine red in colour. When EDTA is added as a titrant, the calcium and magnesium complexes and when all the magnesium and calcium has complexed, the solution turns from wine red in color to blue, marking the end point of the titration. The hardness of the solution is expressed in Eqn 2-14.

$$\text{Hardness (EDTA)as mg CaCO}_3 = \frac{\text{A} \times \text{B} \times 1000}{\text{mL sample}} \quad \textbf{2-14}$$

Where A= mL titration for sample

B= mg $CaCO_3$ equivalent to 1.00 mL EDTA titrant

2.4.2.5 Sediment: Turbidity and Total Suspended Solids

Turbidity was measured using a portable Hach 2100N or 2100P turbidimeter. Total Suspended Solids (TSS) was calculated according to SM2540D (APHA 2005). Glass fibre

composite filter paper was oven dried at 105°C using a Contherm Thermotec 2000 oven and weighed (mass filter paper) on a Sartorius GC1603P analytical balance. The water sample was filtered through the glass fibre filter, dried at 105°C for 1-2 hours, and weighed (mass sample + filter paper). A blank sample was also performed using deionized water instead of the sample (wt. blank). Suspended solids concentration (mg/L) was calculated according to Eqn. 2-15.

$$\text{TSS} = \frac{[(\text{wt sample} + \text{filter paper}) - (\text{wt filter paper}) - (\text{wt blank})] * 1{,}000{,}000}{\text{Sample Volume}} \qquad \textbf{2-15}$$

2.4.2.6 Chemical Oxygen Demand and Dissolved Organic Carbon

Biochemical oxygen demand (BOD) is a measure of the amount of oxygen that bacteria will consume while decomposing organic matter under aerobic conditions. BOD is extremely important to stream health and one of the major regulations placed on wastewater discharges; however, analysis takes five days. While BOD remains an import factor in regulating discharges, other analytical tests such as chemical oxygen demand (COD) and dissolved organic carbon (DOC) are more rapid and easy to perform. Chemical oxygen demand (COD) samples were periodically taken from stormwater effluent and analysed in accordance with Hach (2003) using a Hach DRB 200 spectrophotometer with low range (<150 mg/L) reagents. COD it is a measure of the total quantity of oxygen required to oxidize all organic material (into carbon dioxide and water) simulating ultimate aerobic respiration potential based on carbon concentration. COD is always larger than biochemical oxygen demand (BOD) and thus initial screening of samples using COD can determine if oxygen demand may be an issue in the sampled water. Dissolved organic carbon (DOC) was analysed using a Teledyne Tekma Apollo 9000 Combustion TOC analyser according to SM 5310B (APHA 2005).

2.4.2.7 Microbiology: Faecal Coliforms

Faecal coliforms were sampled and analysed by RJ Hill Laboratories using membrane filtration according to SM9222D (APHA 2005). Faecal coliforms were all below detection limit in the rain garden effluent and thus faecal testing was discontinued.

2.5 Health and Safety Approach

Prior to leaving University of Canterbury (UC) for *Field Related Activities*, mandatory documentation was completed including:

- Hazard Risk Assessment and Management Matrix
- Field Activity Plan
- Field Activity Departure Register
- Activity Leader: Health Declaration and Consent
- Field Activity Participant Consent

Two copies of all paperwork were filed with the Chief Technical Officer and Senior Academic Supervisor in Civil and Natural Resources Engineering. All parties reviewed and were familiar with the documents prior to departing for the field. There were a minimum of two field researchers in the field at all times. All field researchers checked in with the Senior Academic Supervisor upon return.

Chapter 3: Mesocosm-Scale Laboratory Experiments

3.1 Results and Discussion

3.1.1 Untreated (Influent) Stormwater Characterisation

General water quality parameters for untreated stormwater collected from the stormwater feed tank are reported in Table 3-1. Stormwater in the feed tank included homogenized water collected from 16-17 July 2010, 3 September 2010, and 5 November 2010. Untreated stormwater showed a slightly acidic pH (6.23), likely due to mildly acidic precipitation (Pennington and Webster-Brown 2008). The low hardness (21.93 mg/L as $CaCO_3$) and specific conductance (61.93 µS/cm) are comparable to stormwater signatures in the literature, which identified a mean untreated stormwater hardness of 19.28 ± 11.72 mg/L (Engstrom 2004) and mean specific conductance of 113 ± 11 µS/cm (Datry et al. 2003). Turbidity (73.2 NTU) and TSS (98.4 mg/L) showed a moderate level of solids in the stormwater.

Table 3-1- Water quality parameters for untreated stormwater in the stormwater feed tank. Values represent mean ± standard deviation. Specific conductance and pH were continuously logged. (n=8 for hardness, turbidity and TSS samples, n=3 for faecal coliforms) < indicates below detection limit.

Parameter	Untreated Stormwater
pH (S.U.)	6.23 ± 0.08
Hardness (mg/L as $CaCO_3$)	21.61 ± 3.51
Specific Conductance (µS/cm)	61.93 ± 2.28
Turbidity (NTU)	73.2 ± 44.66
TSS (mg/L)	98.4 ± 30.41
Faecal Coliforms (cfu/100 mL)	<10

Mean and median metal (Zn, Pb and Cu) and nitrate concentrations in untreated stormwater collected from the header tank prior to experimental runs are reported in Table 3-2. First flush samples were collected in the field and are presented to show relative magnitude of first flush samples compared with homogenised header tank samples. The median value should be compared with the ANZECC guidance value for discharge compliance. The ANZECC effects-based guidelines are the main thresholds adopted in Christchurch, New Zealand for estimating likely ecotoxicity from discharges into surface water bodies (ECAN 2011). The 90% threshold applies to urban areas, stipulating that at these (median) concentrations, 90% of the species are likely to be unaffected. The guidelines are adjusted for hardness; however, hardness levels in stormwater were low (21.61 mg/L as $CaCO_3$) (Table 3-1), thus no adjustments were required.

Table 3-2- Untreated stormwater concentrations of metals (total and dissolved) and nitrate compared with the ANZECC (2000) surface water trigger values (90%) and drinking water standards, NZDWS (2005). Values represent mean ± standard deviation and median concentrations (µg/L). First flush samples were collected in the first 20 minutes of storm events. (n = 4, 8, 8 for first flush, mean and median analytical results, respectively). NA indicates no exceedance of applicable guidance.

Contaminant		Untreated Stormwater (µg/L)			90% ANZECC Guidance (µg/L)	90% ANZECC Exceedance Factor			New Zealand Drinking-Water Standards (2005) (µg/L)
		First Flush	Mean	Median		First Flush	Mean	Median	
Copper	-Total	33.1 ± 10.6	17.1 ± 3.6	16.6	1.8	18.4	9.5	9.2	2,000
	-Dissolved	5.7 ± 4.7	2.6 ± 0.3	2.6	-	3.2	1.4	1.5	-
Zinc	-Total	302.1 ± 105.0	162.8 ± 28.1	168.0	15.0	20.1	10.9	11.2	1,500[1]
	-Dissolved	96.8 ± 32.9	82.7 ± 18.0	90.5	-	6.5	5.5	6.0	-
Lead	-Total	92.5 ± 25.8	39.8 ± 11.3	35.1	5.6	16.5	7.1	6.3	10
	-Dissolved	6.0 ± 2.3	1.4 ± 0.2	2.3	-	1.1	NA	NA	-
Nitrate		2,540.0 ± 620.0	883.0 ± 126.0	700.0	3,400.0	NA	NA	NA	50,000

Notes: 1. The New Zealand Drinking-water Standards (2005) do not provide a standard for Zinc but give a 1,500 µg/L guidance value.

Median *total* metal concentrations (Cu= 16.6 µg/L, Zn=168.0 µg/L and Pb=35.1 µg/L) greatly exceeded the recommended 90% ANZECC median guidelines by a factor of 9.2 (Cu), 11.2 (Zn) and 6.3 (Pb) (Table 3-2). Exceedance factors increased substantially for the first-flush component in stormwater with total metal exceedance factors of 18.4 (Cu), 20.1 (Zn) and 16.5 (Pb). Comparing total metal concentrations will often overestimate the bioavailable fraction (Landner and Reuther 2004) and thus ANZECC (2000) stipulates to compare the dissolved fraction if total metal concentrations exceed the stipulated trigger values. Median *dissolved* metal concentrations for Cu and Zn exceeded the 90% trigger values by a factor of 1.5 and 6.0, respectively (Table 3-2), highlighting the extent to which untreated stormwater is likely impairing ecological communities within the receiving waterways (Harding 2005).

New Zealand Ministry of Health Drinking Water Standards (NZMOH 2005) are also given (Table 3-2) because if rain garden effluent infiltrates to groundwater used as a water supply, by protocol, the discharge would be subjected to these drinking water standards instead of the surface water ANZECC guidelines. While the drinking water standards are far less stringent than the 90% species protection levels set by ANZECC, concentrations of lead still substantially exceed the drinking water standards (Table 3-2).

It was anticipated that elevated nitrate concentrations might be present in stormwater as reported elsewhere (Taylor et al. 2005; Henderson et al. 2007); however, nitrate concentrations measured in the stormwater (883 µg/L) were less than the 90% ecotoxicological threshold value of 3,400 µg/L and much less than the drinking water standard of 50,000 µg/L (Table 3-2).

3.1.2 Hydraulic Performance

Saturated hydraulic conductivity, a measure of the infiltrative capacity, varied for each of the three different systems. Measurements collected before (initial) and after (final) the water quality treatment experiments were similar, with no apparent indication of system clogging (Table 3-3).

Table 3-3- Hydraulic parameters of mesocosm-scale rain gardens prior to (initial) and following (final) contaminant removal experiments. Values represent mean ± standard deviation. Ranges are shown in parenthesis below the mean. (n =25)

Hydraulic Conductivity (K_{sat})	Sand	Sand/Topsoil	Topsoil
Initial (mm/hr)	805.4 ± 4.2 (789.7 - 810.5)	302.1 ± 4.6 (285.8 - 309.7)	163.7 ± 2.9 (160.2 - 171.6)
Final (mm/hr)	799.9 ± 4.6 (789.7 - 808.8)	290.4 ± 4.6 (281.0 - 296.8)	159.8 ± 1.6 (156.5 - 162.2)

Mean hydraulic conductivity ranged from 805.4 mm/hr (initial) in the sand-only system to 159.8 mm/hr (final) in the topsoil-only system (Table 3-3). Hydraulic conductivities observed for sand and sand/topsoil systems were consistent with sandy gravel, and hydraulic conductivities in topsoil systems were consistent with silty sand (Mays 2004). This range of hydraulic conductivities equates to an order of magnitude greater than the minimum allowable conductivity of 13 mm/hr stipulated in California (SFPUC 2009), Maryland (MDE 2009) and New Zealand (ARC 2003) rain garden design guidelines (Table 1-3).

System design was approached from a conservative perspective (with regard to hydraulic conductivity), to ensure the systems did not clog. The uppermost layer of bioinfiltration systems is typically subject to clogging due to suspended sediment from stormwater (Hatt et al. 2008; NZTA 2010). Most international stormwater design manuals therefore recommend pre-treatment (i.e. sumps, filter strip, etc.) to remove the heavier particulates prior to filtration in the bioinfiltration zone (ARC 2003; GVRD 2005; BCC 2007; MDE 2009). In this study, the stormwater feed tank effectively functioned as a pre-treatment sump removing the larger solids which were allowed to settle for 5 minutes before the stormwater was applied to the experiments systems (Section 2.1.4.3). This may explain why there was no apparent difference in saturated hydraulic conductivity in the systems following the water quality treatment experiments (Table 3-3).

3.1.3 Real-Time Water Quality Monitoring

Real-time water quality monitoring of pH, specific conductance, nitrate, and temperature (Section 2.3.2) was also performed on effluent from each of the mesocosm-scale rain garden systems throughout experimental runs (three runs). Figure 3-1, Figure 3-2, and Figure 3-3 show trends in pH, specific conductance and nitrate, respectively, over three runs from each

of the three systems (sand, sand/topsoil, and topsoil). (Temperature was recorded only to account for adjustment to pH due to temperature change and to normalize electrical conductivity to 25°C (specific conductance).) A summary table of pH, specific conductance, and nitrate is shown in Table 3-4. The length of logging was different for each system (sand up to 96 mins, sand/topsoil up to 164 mins, and topsoil up to 236 mins) due to different hydraulic conductivities, and therefore took different periods to drain to field capacity.

Table 3-4- Water quality in raw stormwater (inflow) and outflow (i.e. treated effluent) from all experimental systems at 'standard' loading. (Cu= 5.99 ± 0.73 µg/min, Zn= 57.89 ± 6.06 µg/min, Pb= 13.65 ± 2.80 µg/min).

Parameter	Mean Inflow ± SD	Mean Effluent ± SD		
	Raw Stormwater	Sand	Sand/Topsoil	Topsoil
pH (SU)	6.23 ± 0.08	7.38 ± 0.03	6.60 ± 0.04	6.24 ± 0.08
Specific Cond. (µS/cm)	62 ± 2	112 ± 16	152 ± 28	139 ± 21
Nitrate (µg/L)	883 ± 126	3,610 ± 1,600	4,560 ± 1,106	4,330 ± 1,317
n	8	96	164	236

Three distinct pH trends, one from each system, are evident (Figure 3-1). The sand only system (Sand R1, Sand R2 and Sand R3) maintained a much higher pH (7.38 ± 0.03) than the topsoil-only system (pH 6.24 ± 0.08). This may be explained by an alkaline buffering effect afforded by the calcareous content within the sand system, which maintained a pH of approximately one SU above influent stormwater (pH 6.23 ± 0.08). Conversely, the topsoil-only system remained at pH 6.24 ± 0.08, approximately the same as the influent stormwater pH across all three experimental runs since it lacked an alkalinity source such as sand. The sand/topsoil system was afforded less alkalinity buffering than the sand-only system, but still maintained an elevated pH of 6.60 ± 0.04 compared with influent stormwater.

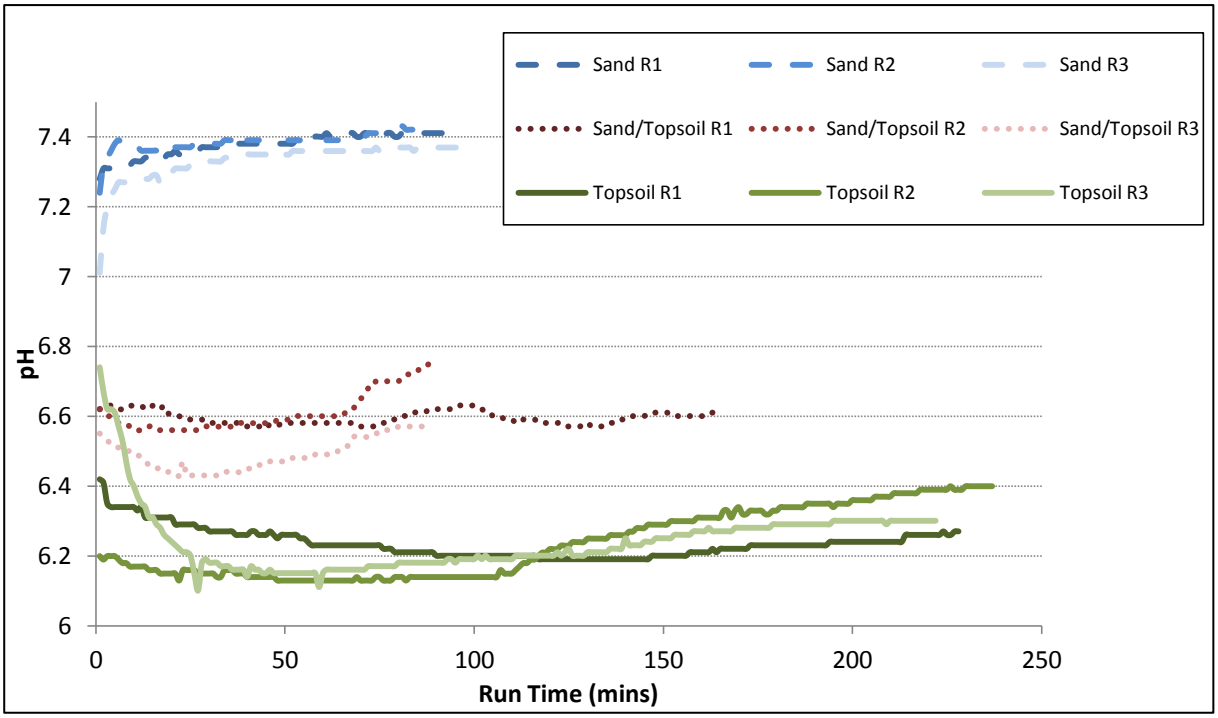

Figure 3-1- Continuous pH monitoring in mesocosm-scale laboratory rain garden effluent. Values (SU) were recorded every minute throughout experimental runs. Runs are shown separately and identified with (R1), (R2) and (R3) corresponding to run 1, run 2 and run 3, respectively.

Real-time logging of specific conductance is shown in Figure 3-2. Specific conductance, a measure of the (temperature-normalised) total ionic concentration of the liquid, revealed less discernable trends between treatments as compared with pH. Specific conductance in all system effluents was elevated compared to the influent (62 ± 2 µS/cm) across all experimental runs (sand 112 ± 16 µS/cm, sand/topsoil 152 ± 28 µS/cm, topsoil 139 ± 21 µS/cm). This increase is likely due to the release of ions from the organic material (bark mulch, topsoil). Specific conductance measured in system effluent was comparable with other BMPs (average 209 ± 400 µS/cm (Pan and Duan 2011). A comprehensive review of 114 BMP sites found a similar increase in specific conductance at effluent as compared with untreated stormwater (Pan and Duan 2011).

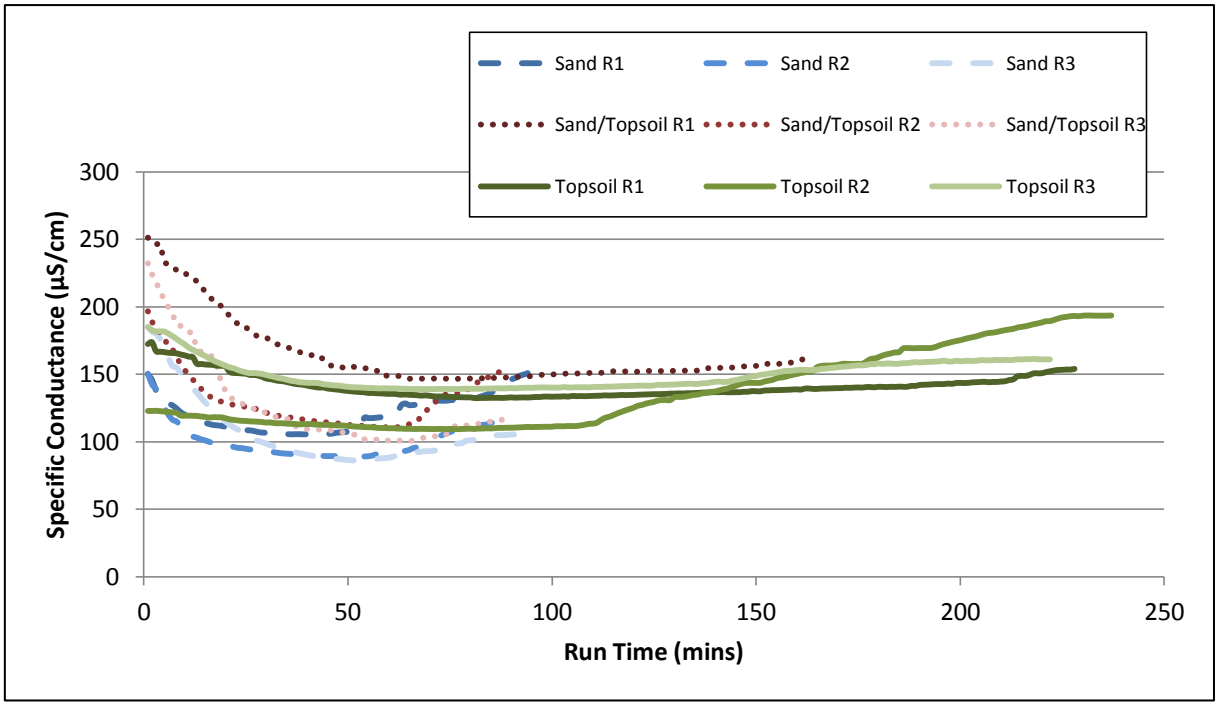

Figure 3-2- Continuous specific conductance monitoring in mesocosm-scale laboratory rain garden effluent. Values (µS/cm) were recorded every minute throughout experimental runs. Runs are shown separately and identified with (R1), (R2) and (R3) corresponding to run 1, run 2 and run 3, respectively.

Nitrate (NO_3^-) concentrations, likely originating from the topsoil and/or bark (Section 3.1.7), were higher in the effluent (from all systems) compared to the influent (Table 3-4). Average concentrations in topsoil and sand/topsoil systems effluent (4,330 and 4557µg/L, respectively) were higher than the sand-only effluent of 3,613 µg/L (Table 3-5, Figure 3-3). Mean influent nitrate concentration was 883 µg/L with a first flush concentration of 2,540 µg/L (Table 3-2). Median nitrate concentrations were also compared as they are most applicable for direct comparison with relevant 90% ANZECC guidelines (Table 3-5). Influent median nitrate concentrations of 700 µg/L were lower than the effluent for sand, sand/topsoil, and topsoil of 3,000, 4,900, and 4,200 µg/L, respectively, with systems containing topsoil exceeding the ANZECC guidelines (Table 3-5). Similar nitrate leaching from non-vegetated biofiltration systems was observed in research performed by Henderson et al. (2007), with leached concentration up to 2,540 µg/L compared with 690 µg/L in influent. Alternate wet/dry cycles can affect nitrate leaching from soils (Randall and Mulla 2001), and rain gardens undergo such cycles due to the sporadic nature of stormwater. Since the mesocosm laboratory stormwater systems were not vegetated, they were not afforded the same opportunity for nitrate immobilization compared with vegetated bioinfiltration systems.

Table 3-5- Median nitrate concentration in system effluent. (n= 96, 164, and 236 for sand, sand/topsoil, and topsoil system effluent, respectively)

Median Values (µg/L)	90% ANZECC Guidance (µg/L)	Outflow		
		Sand	Sand/Topsoil	Topsoil
Nitrate	3,400	3,000	4,900	4,200

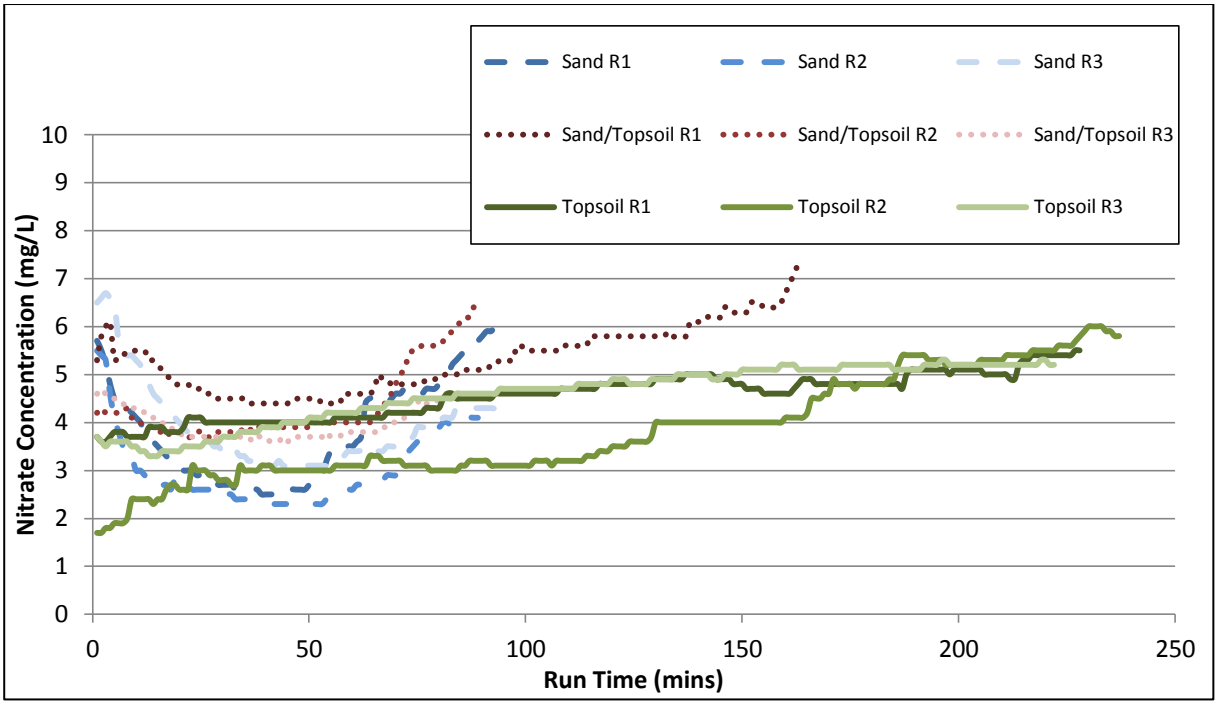

Figure 3-3- Continuous nitrate monitoring in mesocosm-scale rain garden effluent. Values (mg/L) were recorded every minute throughout experimental runs. Runs are shown separately and identified with (R1), (R2) and (R3) corresponding to run 1, run 2 and run 3, respectively.

3.1.4 Contaminant Removal Efficiency

3.1.4.1 Effluent Concentrations

Concentrations of total and dissolved metals from effluent in all laboratory rain gardens subjected to the standard contaminant loading rate (Cu: 5.99 ± 0.73 µg/min, Zn: 57.89 ± 6.06 µg/min and Pb: 13.65 ± 2.80 µg/min) were compared with influent concentrations and the relevant ecotoxicological ANZECC values (ANZECC 2000) (Table 3-6).

Table 3-6- Contaminant concentrations in untreated stormwater (inflow) and outflow (i.e. treated effluent) from all experimental systems at 'standard' loading compared with relevant 90% ANZECC guidance. Standard loading (Cu= 5.99 ± 0.73 µg/min, Zn= 57.89 ± 6.06 µg/min, Pb= 13.65 ± 2.80 µg/min). Values represent median and mean ± standard deviation contaminant concentrations (µg/L). Percent dissolved metal fraction was calculated from mean concentrations. (n=8 for inflow and 27-56 for effluent)

Contaminant (µg/L)		ANZECC Guidance (90%)	Inflow			Effluent								
			Untreated Stormwater			Sand			Sand/Topsoil			Topsoil		
			Median	Mean	± SD	Median	Mean	± SD	Median	Mean	± SD	Median	Mean	± SD
Copper	Total	1.8	16.6	17.1	± 3.6	7.9	12.0	± 11.1	11.4	10.7	± 2.3	22.7	37.9	± 34.8
	Dissolved		2.6	2.6	± 0.3	2.0	2.9	± 3.6	6.3	6.0	± 1.5	16.7	30.2	± 32.4
	% Dissolved			15%			24%			56%			80%	
Zinc	Total	15.0	168.0	162.8	± 28.1	47.2	76.6	± 125.6	42.6	51.2	± 57.3	112.3	149.5	± 96.1
	Dissolved		90.5	82.7	± 18.0	2.4	23.1	± 93.6	22.8	27.7	± 31.3	105.1	141.0	± 95.3
	% Dissolved			51%			30%			54%			94%	
Lead	Total	5.6	35.1	39.8	± 11.3	6.7	11.6	± 14.8	5.3	6.1	± 5.0	8.7	10.1	± 6.3
	Dissolved		2.3	1.4	± 0.2	0.1	0.2	± 0.1	0.6	0.5	± 0.3	3.7	3.5	± 1.6
	% Dissolved			4%			1%			9%			35%	

All total mean and median metal (Zn, Pb and Cu) concentrations in the effluent were below their respective influent values showing contaminant removal within each system (Table 3-6). An exception was for Cu from the topsoil-only system, which had an average effluent concentration of 37.9 µg/L compared with the raw influent of 17.1 µg/L. The effluent Cu concentration increased as the run progressed and effluent flow decreased (Figure 3-4), resulting in a higher average effluent value. Individual (time-step) contaminant loading calculations (a product of concentration and flow) confirmed that Cu did not leach from the substrate on any particular occasion. Zn and Pb concentrations showed similar trends to Cu in terms of increased concentration at low flow (Figure 3-5). Clearly, the system containing only topsoil was not conducive to retaining Cu or Zn as well as the systems containing sand. This may be explained by the enhanced pH buffering afforded in the sand discussed earlier (Section 3.1.3). Furthermore, the topsoil-only system showed the highest increase in concentration at low effluent flow rate for all metals (Figure 3-4).

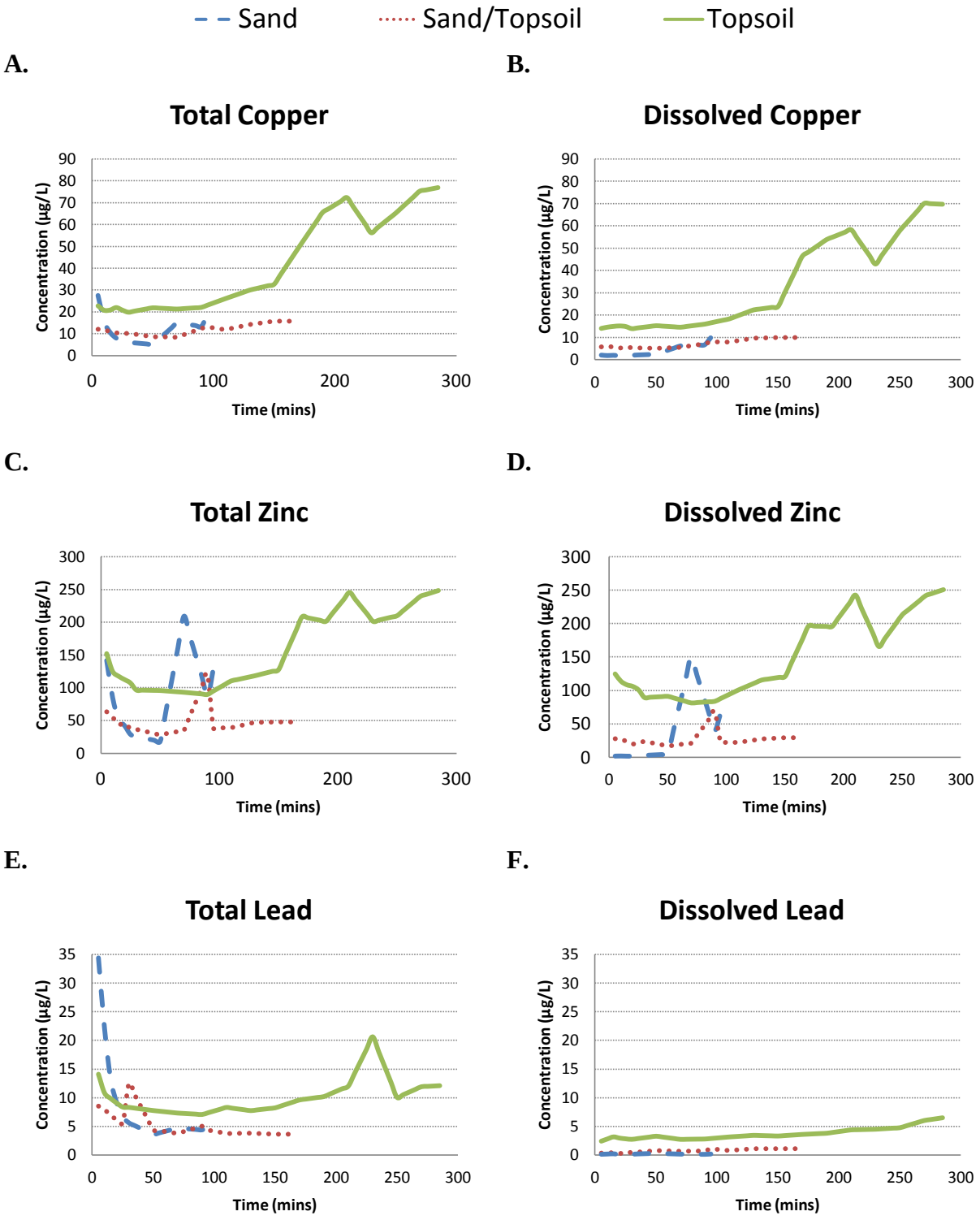

Figure 3-4- Metal (total and dissolved) effluent concentration (µg/L) over time at 'standard' loading. Mean influent concentration total Cu= 16.96 ± 2.07 µg/L, dissolved Cu = 2.66 ± 0.23 µg/L, total Zn= 163.79 ± 17.15 µg/L, dissolved Zn = 85.52 ± 15.13, total Pb= 38.61 ± 7.92 µg/L, dissolved Pb = 1.66 ± 1.05 µg/L.

With the exception of Pb from the Sand/Topsoil system, effluent values were in exceedance of the 90% guidance trigger values and thus may be considered an ecological threat to aquatic ecosystems if discharged without further treatment. The extent to which the effluent concentrations were in exceedance of the 90% ANZECC trigger values varied (Table 3-7). Total Cu exceedance factors were highest, ranging from 4.4 (sand) to 12.6 (topsoil), while total Zn exceedance factors ranged from 2.8 (sand/topsoil system) to 7.5 (topsoil). Total Pb had lower exceedances at 1.2 (sand-only) and 1.5 (topsoil). Dissolved fractions were also compared with the ANZECC guidance, as dissolved metal species are more bioavailable and thus a better indication of ecotoxicity (Depledge et al. 2009). Dissolved concentrations of Cu and Zn exceeded relevant trigger values. Although effluent concentrations are reduced from influent values, effluent concentrations remain above the ecotoxicological trigger values and thus could negatively impact the receiving surface water environment. Comparing system effluent directly with ecotoxicological trigger values is a conservative approach compared with estimating potential concentrations at the edge of the mixing zone, which cannot accurately be determined in a laboratory setup and will vary considerable between sites.

Table 3-7- Contaminant exceedance factors of ANZECC guidance in untreated stormwater (inflow) and outflow (i.e. treated effluent) from all experimental systems at 'standard' loading. Exceedance factors are calculated from median concentrations (See Table 3-5). NA indicates there was no exceedance of ANZECC guidance. (n=8 for inflow and 27-56 for effluent)

Contaminant		ANZECC Guidance (90%) µg/L	ANZECC (90%) Exceedance Factors			
			Inflow	Effluent		
			Raw Stormwater	Sand	Sand/Topsoil	Topsoil
Copper	Total	1.8	9.2	4.4	6.3	12.6
	Dissolved		1.4	1.1	3.5	9.3
Zinc	Total	15.0	11.2	3.1	2.8	7.5
	Dissolved		6.0	NA	1.5	7.0
Lead	Total	5.6	6.3	1.2	NA	1.5
	Dissolved		0.4	NA	NA	NA

3.1.4.2 Contaminant Load

Contaminant effluent load calculations were computed from flow rate measurements collected throughout experimental runs and contaminant concentrations measured at the same time (Figure 3-5). Understanding total contaminant mass removal in treatment systems is important for determining their treatment capacity (i.e. estimated longevity) and hence sizing requirements. Treatment systems feasibility must evaluate influent stream characteristics, mass loading rates and desired treatment goals (Sansalone 1999; McCauley et al. 2009).

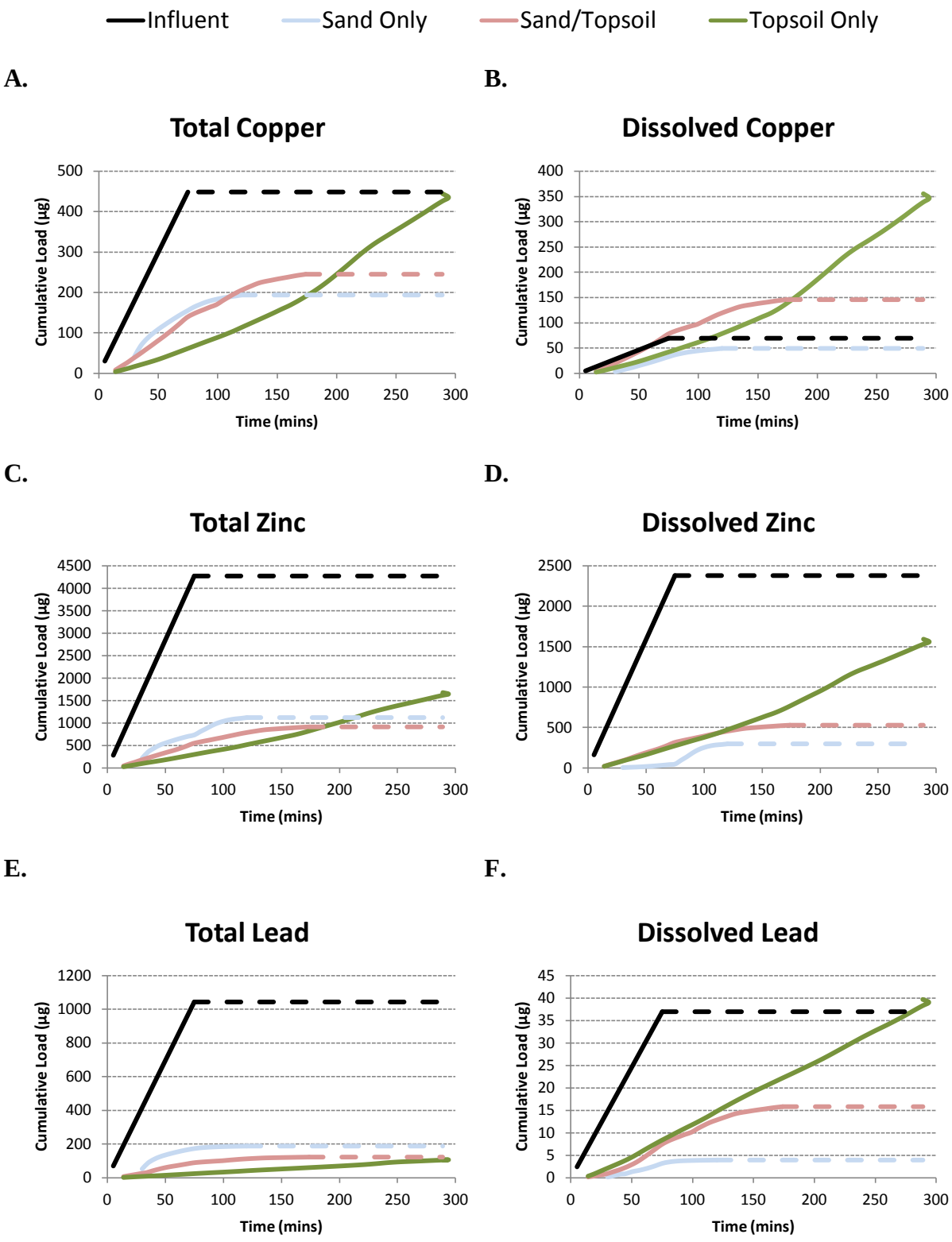

Figure 3-5- Cumulative influent and effluent metal (total and dissolved) load over time at 'standard' contaminant loading. Standard loading (Cu= 5.99 ± 0.73 µg/min, Zn= 57.89 ± 6.06 µg/min, Pb= 13.65 ± 2.80 µg/min). Dashed lines indicate stormwater was no longer being applied/discharged from system and thus cumulative contaminant load remains constant.

Metal mass loading (μg) over time (minutes) computations show that low cumulative (total) Zn (Figure 3-5C) and Pb (Figure 3-5E) amounts were discharged compared with the influent metal loads, confirming metal removal for all systems throughout the experimental run times. Although stormwater was only applied to each system for 73 minutes (Section 2.1.7), effluent continued to discharge from each system for much longer (sand up to 96 mins, sand/topsoil up to 164 mins, and topsoil up to 236 mins) because of the different hydraulic conductivities and water retention capacity in each system (Section 3.1.2). Therefore, there was a short lag-time between when stormwater was applied to the systems and when it drained from them. Removal of total Cu, Zn and Pb was greater than 50% for all systems, except Cu in the topsoil-only system that had a negligible reduction (0.3%) due to a high dissolved fraction (Figure 3-5B). The consistent total metal removal throughout the experimental runs offers promising results in terms of treatment efficiency for stormwater bioinfiltration systems. These data contribute to the dearth of knowledge in the literature on bioinfiltrative system performance.

The pollutant removal trends were very different for the topsoil-only system compared with the other two treatments (Figure 3-5). For Cu and Zn, cumulative total metal removal (Figure 3-5B and D) in the sand and sand/topsoil systems was always greater compared to the topsoil-only system reflecting the enhanced pH buffering afforded in the sand (Reynolds et al. 1986; Plassard et al. 2000) that resulted in an elevated pH which promoted particulate species accounting for better metal removal (Pitcher et al. 2004; Sansalone and Glenn 2007). Cumulative dissolved Cu amounts in the topsoil and sand/topsoil systems (Figure 3-5B) were greater than their actual application load, which was attributed to a change in Cu speciation from particulate to dissolved fractions. Dissolved fractions are promoted below pH 7 (Dempsey et al. 1993), which are not as effectively removed in biofiltration systems (Fletcher et al. 2004), and thus can be net-exported. This was particularly pronounced for Cu, where dissolved fraction (Figure 3-5B) was a large percentage of cumulative metal load (Figure 3-5A).

Metal removal as a function of metal speciation was further examined by plotting the relative proportions of particulate and dissolved fractions that were removed in each system (Figure 3-6). It was originally hypothesised that the topsoil-only system could demonstrate better metal removal due to due to a high affinity for metal sorption to soil (Davis et al. 2001).

However, the converse was observed for Cu and Zn (Figure 3-6). For instance, effluent discharged from the topsoil system contained 4,973 µg Cu/m^3 substrate (0.3% removed) and 18,785 µg Zn/m^3 substrate (60.5% removed), while effluent from the sand system contained 2,174 µg Cu/m^3 substrate (56.4% removed) and 12,589 µg Zn/m^3 substrate (73.5% removed). Furthermore, the topsoil-only system had the largest proportion of dissolved metals (80% Cu and 94% Zn) compared to the sand-only system (24% Cu and 30% Zn) or sand-topsoil mix (56% Cu and 54% Zn) (Figure 3-6), highlighting the effect of topsoil on metal speciation in stormwater bioinfiltration systems within the observed pH range (6.23-7.38) (Section 3.1.3). The effect was most pronounced for Cu; however changes in speciation in Zn, and to a much lesser extent Pb, were also apparent. The poor removal of copper and high dissolved fraction is likely in part due to copper complexes with organic ligands (dissolved forms) as reported in the literature (Mason et al. 1999; Landner and Reuther 2004). Since Pb is known to be more dominant in the particulate form, it is removed in bioinfiltration systems much easier, accounting for the high removal efficiency (Figure 3-6).

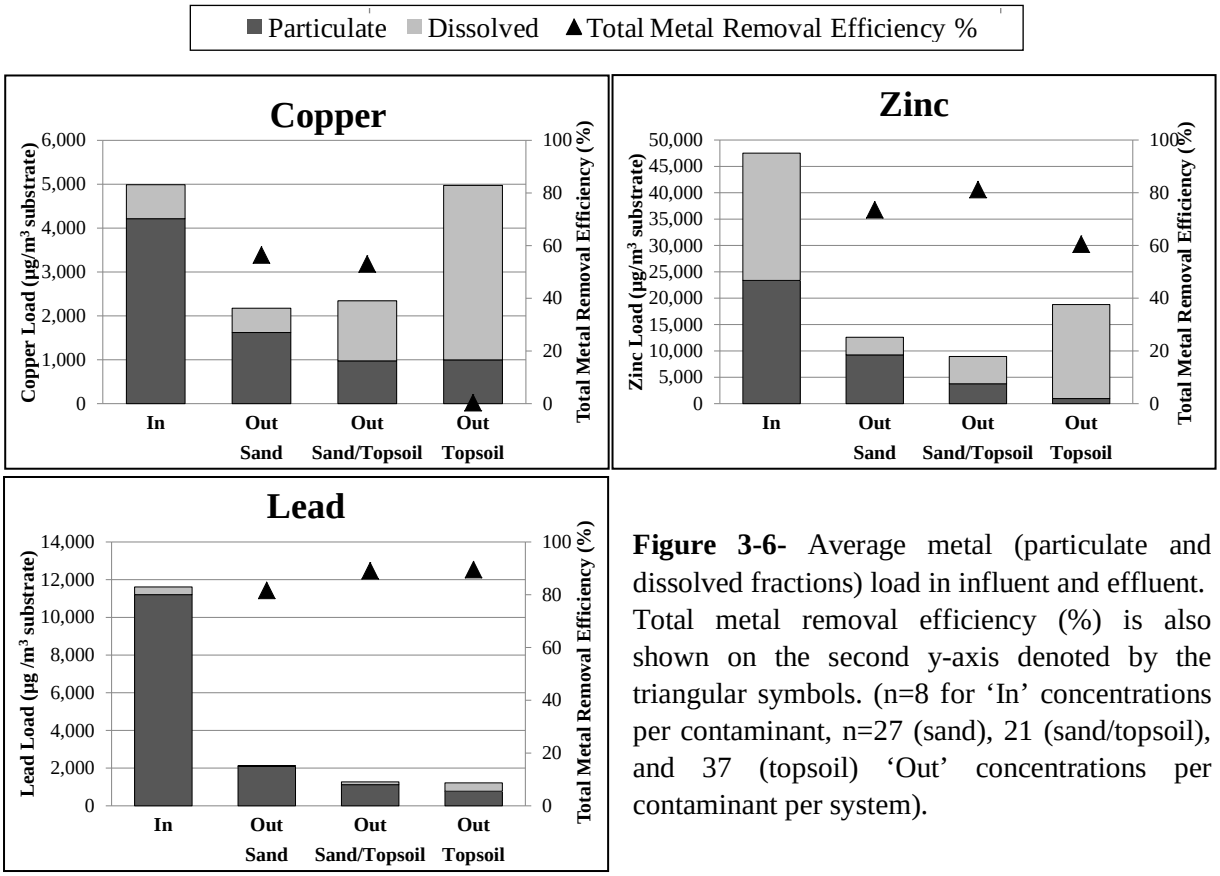

Figure 3-6- Average metal (particulate and dissolved fractions) load in influent and effluent. Total metal removal efficiency (%) is also shown on the second y-axis denoted by the triangular symbols. (n=8 for 'In' concentrations per contaminant, n=27 (sand), 21 (sand/topsoil), and 37 (topsoil) 'Out' concentrations per contaminant per system).

3.1.4.3 pH Effects

To further investigate the apparent relationship between effluent pH and dissolved metal fractions, regression relationships were derived for each of the dissolved metals with pH in the effluent (Figure 3-7).

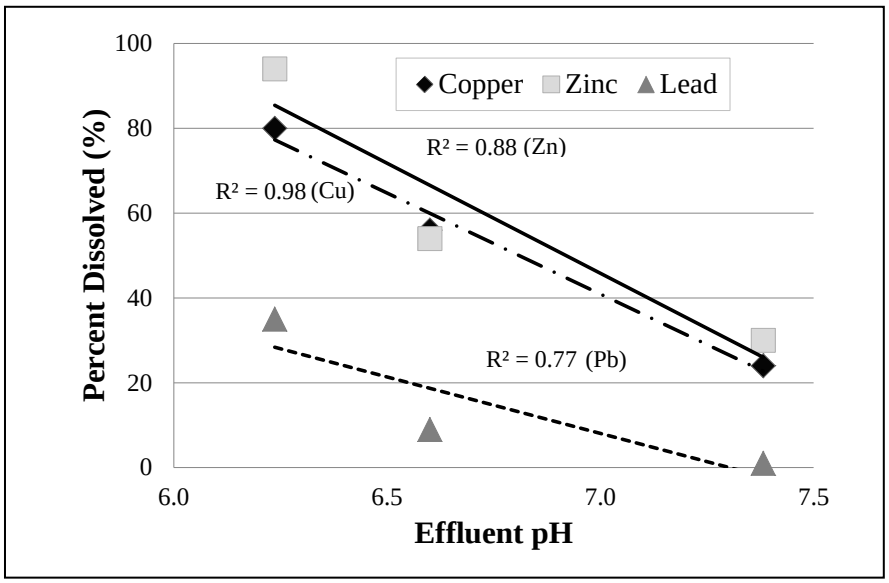

Figure 3-7- Metal regression trends as a function of stormwater effluent pH. Each symbol represents the average percent dissolved at average effluent pH. n=76 per metal.

The regression lines provide a general indication of the relationship of dissolved metals and pH. The correlation between pH and dissolved fraction was highly significant (Pearson's Correlation, $p < 0.0001$, $d_f = 74$) for Cu, Pb, and Zn. Results indicate that pH (within the range of influent stormwater (6.23) to effluent sand-only system (7.38)) influences metal speciation, and hence, metal removal capacity in bioinfiltration systems. The effluent pH from the sand and sand/topsoil systems was somewhat raised to 7.38 and 6.60, respectively, compared with the raw stormwater pH of 6.23 (Table 3-4). This likely resulted from carbonate in the sand component (Reynolds et al. 1986; Plassard et al. 2000), while the topsoil-only pH was not buffered and remained at 6.24, resulting in poorer metal treatment. The partitioning between the particulate-bound and dissolved forms shifts towards the dissolved species below approximately pH 7, as particulate-bound trace metals are released from particles as free ions (Dempsey et al. 1993; Engstrom 2004). While it is well reported that pH influences metal speciation (Pitcher et al. 2004; Sansalone and Glenn 2007), this phenomenon is not currently considered in any stormwater treatment design guidelines. However, this is an especially important phenomenon that should be considered for most stormwater treatment systems in New Zealand whose water quality objective (ARC 2003) is based on the premise that metals (i.e. including Zn and Cu) are removed with concurrent removal of 75% TSS (Smythe et al. 2007) – an assumption that is incorrect since metals can clearly prevail in dissolved states within the pH range typical of stormwater. Without a pH amendment, rain gardens (and other filtration systems relying on particulate removal) are unlikely to provide adequate total metal removal if the stormwater pH is even slightly acidic.

3.1.4.4 Standard vs. High Metal Loading Rates

Removal efficiencies for each system under 'standard' and 'high' metal loading rates were also measured to assess the treatment capacity of the systems to remove metals at enhanced contaminant loading regimes. Higher loading regimes are commonly used to assess maximum removal capacity of laboratory systems. This higher loading rate enabled the systems to be tested under a greater stormwater load than is typically afforded in field monitoring studies within the timeframe of a research degree. Lead removal was very good across all systems (ranging between 81.6% for 'standard' loading up to 98.6% (Table 3-8) for 'high' (i.e. double) loading). Zn and Cu were generally removed better in the sand-only system than the topsoil-only system. For instance, during application of the high loading rate, 94.5% Zn was

removed in the sand system but only 71.4% in the topsoil system, while Cu removal was 83.3% in the sand system and 69.0% in the topsoil system (Table 3-8). Overall, removal of all metals was greater at the higher loading rate indicating that the systems have not yet reached treatment capacity; however, greater metal removal at higher loading does not imply effluent concentrations are below that of standard loading. The limited data on field rain gardens indicates that performance might be compromised after a 5-year period likely due to diffuse sediment input clogging the filtration media (Galli 1992).

Table 3-8- Total metal removal efficiency (expressed as a %) during 'standard' and 'high' contaminant loading rates. Standard loading (Cu = 5.99 ± 0.73 µg/min, Zn = 57.89 ± 6.06 µg/min, Pb = 13.65 ± 2.80 µg/min), high loading (Cu = 11.35 ± 2.96 µg/min, Zn = 91.51 ± 23.41 µg/min, Pb = 30.61 ± 9.08 µg/min) (n=3-8 for influent, concentrations reported as mean ± standard deviation)

		Standard Loading			High Loading		
		Sand	Sand/Topsoil	Topsoil	Sand	Sand/Topsoil	Topsoil
Cu	-Influent Conc. for all systems (µg/L)		17.1 ± 3.6			32.1 ± 8.4	
	-Removal Efficiency (%)	56.4	53.0	0.3	83.3	77.4	69.0
Zn	-Influent Conc. for all systems (µg/L)		162.8 ± 28.1			258.9 ± 66.2	
	-Removal Efficiency (%)	73.5	81.2	60.5	94.5	87.9	71.4
Pb	-Influent Conc. for all systems (µg/L)		39.8 ± 11.3			86.8 ± 25.7	
	-Removal Efficiency (%)	81.6	89.1	89.5	97.3	96.9	98.6

Cumulative total and dissolved metal effluent loads over time were calculated for the higher loading regime to discern differences in metal speciation at this higher loading rate (Figure 3-8). While total Zn, Cu and Pb removal was very good for all systems (Table 3-8, Figure 3-8A, C, E), removal of the dissolved fractions were not (Figure 3-8B, D, F). It is interesting to observe that dissolved Cu amounts in the effluent of the topsoil and sand/topsoil systems were higher than dissolved Cu amounts in their influent (Figure 3-8B), indicating that particulate Cu was mobilised to dissolved species during bioinfiltration at both standard (Section 3.1.4.1) and higher contaminant loading regimes.

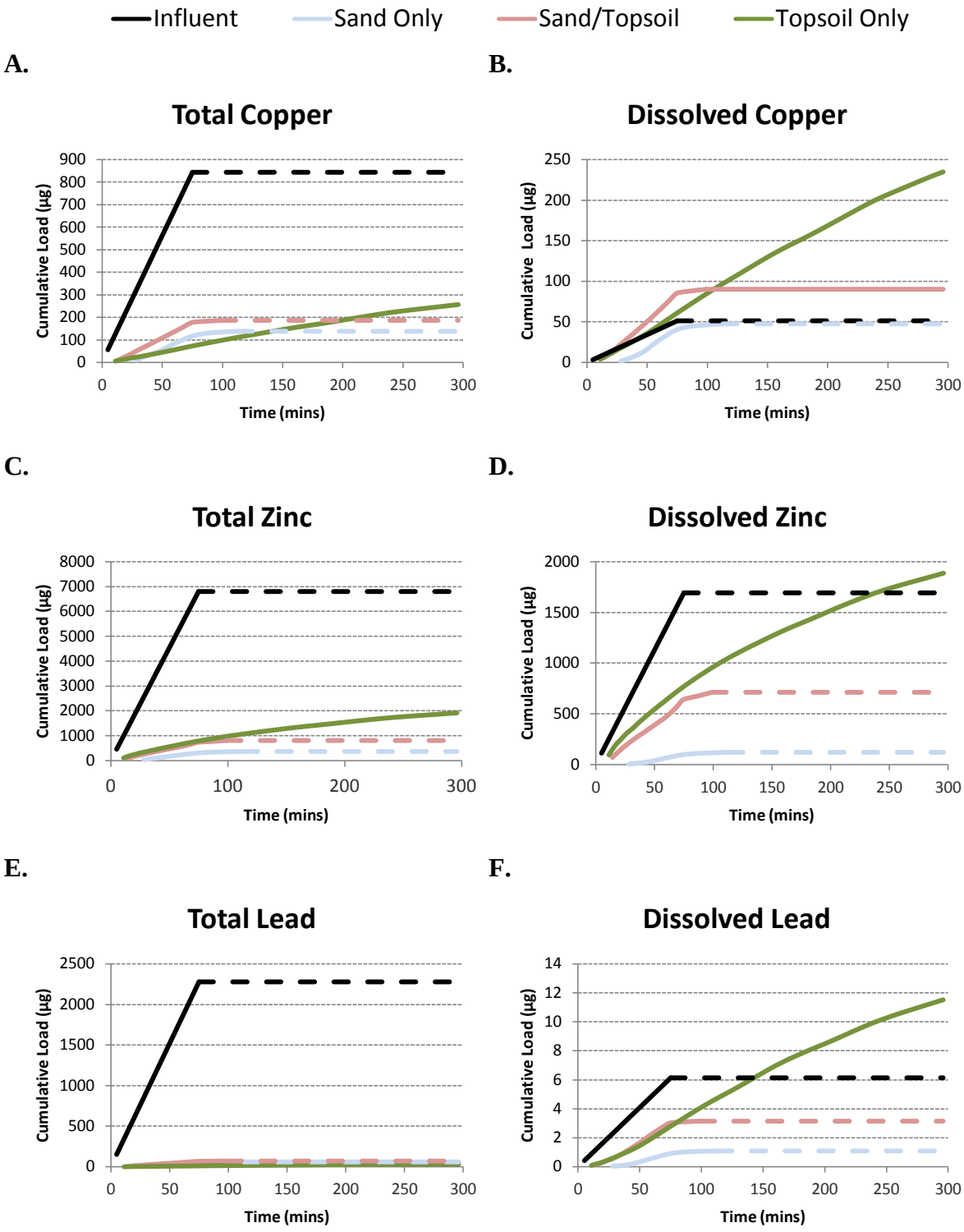

Figure 3-8- Average cumulative metal (total and dissolved) influent and effluent load over time during 'high' contaminant loading regime. Dashed lines indicate stormwater was no longer being applied/discharged from system and thus cumulative contaminant load remains constant. High loading (Cu= 11.35 ± 2.96 µg/min, Zn= 91.51 ± 23.41 µg/min, Pb= 30.61 ± 9.08 µg/min).

3.1.5 Additional Water Quality Parameters

Additional water quality parameters were measured in effluent from the treatment experiments (Table 3-9) since these can provide an indication of the overall water quality discharging from the treatment systems, which can affect health of the receiving waterways. Faecal coliforms were below detection limit all system effluent and not further analysed in this research.

Table 3-9- Additional water quality parameters in laboratory mesocosm-scale system effluent. (n= 6 for DOC and COD and 27-56 for turbidity and hardness, n=1 for faecal coliforms) < indicates below detection limit

	Mean Effluent from Treatment System ± SD		
Contaminant	**Sand**	**Sand/Topsoil**	**Topsoil**
DOC (mg/L)	1.6 ± 1.6	11.5 ± 3.5	15.3 ± 3.2
COD (mg/L)	50.5 ± 6.1	105.0 ± 10.4	134.0 ± 11.0
Turbidity (NTU)	386.9 ± 42.9	136.2 ± 22.4	124.9 ± 18.1
Hardness (mg/L as $CaCO_3$)	36.1 ± 8.3	24.5 ± 3.5	20.0 ± 3.4
Faecal Coliforms (cfu/100 mL)	<100	<100	<100

DOC is an indication of the total bioavailable carbon concentration in solution. It is believed that DOC can be important in reducing the toxicity of dissolved metals through complexation (Santore et al. 2001). In the systems containing topsoil, DOC was higher in the effluent at 11.5-15.3 mg/L most likely because the organic component of these systems was much higher than in the sand-only system of 1.6 mg/L (Table 3-9).

COD concentrations increased with increasing topsoil substrate (Table 3-9) and are elevated above average biofiltration effluent (42.54 mg/L) reported by (Pan and Duan 2011)) who compiled data from 7 different biofiltration systems. Elevated COD levels are likely due to an initial discharge of labile organics as a result of the “newness” of the systems. Once systems reach equilibrium, their COD levels are expected to decrease to levels similar to those identified in the literature.

Turbidity is a measure of the cloudiness of the water and can be detrimental to receiving waterways, as suspended material can prevent light from penetrating deeper into the water, in turn reducing photosynthetic rates and the production of oxygen. Turbidity is also an indicator for TSS/particles that might settle down in a stream bed and negatively impact stream health.

Turbidity was significantly elevated in the sand system (386.9 NTU), which was constructed from unwashed sands. Despite extensive system flushing prior to the start of the laboratory experiments, turbidity levels in effluent from the sand system remained high.

Hardness is extremely important in water quality due to its impact on contaminant toxicity. At increased hardness levels, Ca^{2+} and Mg^{2+} compete with free metal ions for complexing sites, thus reducing metal toxicity (Hyne et al. 2005). Hardness was slightly elevated in the sand system (36.1 mg/L as $CaCO_3$) and decreased with decreasing sand content in the Sand/Topsoil system (24.5 mg/L as $CaCO_3$). This increase in hardness is likely due to carbonate content in the unwashed sand. Hardness concentrations were low compared with the literature, which reported average effluent concentrations (53 sites) of 77.49 ± 139.21 mg/L as $CaCO_3$ (Pan and Duan 2011).

Samples for TSS were also collected; however they were not analysed because of the extended holding times exceeding the QA/QC procedures per APHA Method 2540 resulting from the major earthquake in Christchurch on 22 February at the time of the experiment.

3.1.5.1 TSS-Turbidity Relationships

A correlation between TSS and turbidity can provide a useful indication of the relationship between these two parameters once it is established for characterised wastewater signatures. Turbidity (instantaneous measurement) can be correlated to total suspended solid (TSS) concentrations (laboratory measured) and can be a useful surrogate parameter of suspended solids in stormwater applications (Thomson et al. 1997). Even though TSS data from the treatment experiments did not eventuate due to the major earthquake disruption at the time of the experiments, TSS effluent samples were collected after the removal efficiency experiments following tap water applications (of 0.35 L/min - the same rate as treatment experiments) in order to ascertain the TSS-Turbidity relationship. Effluent turbidity values between the treatment experiments using stormwater and the post-treatment experiments employing tap water were also compared for validity purposes. Pearson correlation coefficients (R values) between for the relationships showed there was a strong relationship between turbidity and TSS in each of the systems (Figure 3-9). The turbidity-TSS R^2 value for sand (0.95) was greater than both the sand/topsoil (0.92) and topsoil (0.80) systems. This

higher correlation may be due to large range of turbidity values recorded in the sand effluent, evidenced by the higher standard deviation in Table 3-9.

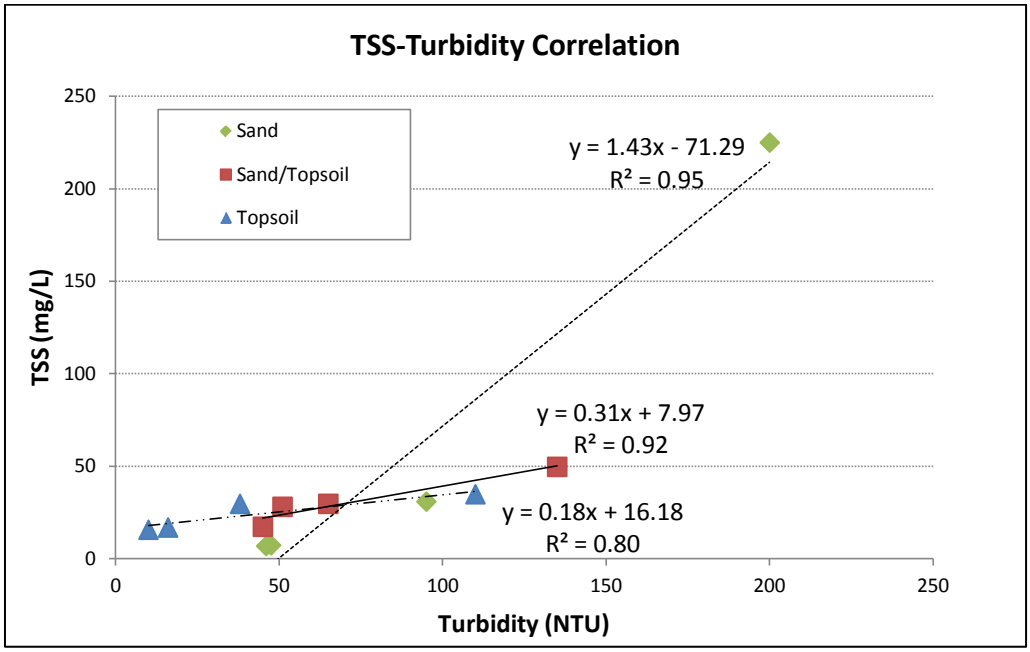

Figure 3-9- TSS- Turbidity correlation relationships as a function of substrate. (n = 4)

3.1.6 Post-Experimental Substrate Autopsies

Substrate samples were collected following the completion of the treatment efficiency experiments outlined earlier (Section 2.1.3.1) in order to estimate a mass balance for metals within each system. Since none of the three metals (Zn, Pb and Cu) have gaseous forms, and the systems were not vegetated, any difference between total (i.e. cumulative) mass influent loading and total mass in the effluent over the course of all experimental runs was assumed to have been retained within the substrates. Since the systems were not actually running for a period of years (despite the higher loading regimes applied) compared with field systems, which are designed for > 10 year life before substrate requires replenishment (Urbonas 1999; Davis et al. 2003; Hatt et al. 2008), large metal concentrations in the substrates were not anticipated. Additionally, any elevated substrate metal concentrations were expected to be measured in the upper surface layers as reported elsewhere (Hatt et al. 2008).

Metal concentrations in the substrates (collected from the top 50 mm per Section 2.3.1) were compared with the Interim Sediment Quality Guidelines (ISQG), which are appropriate for these metals in New Zealand (McCready et al. 2006). All metal concentrations were below

the high and low ANZECC ISQG indicating metal toxicity is not a cause for concern in these systems yet (Table 3-10). Slightly elevated copper (7.57 mg/kg dry weight) and zinc (54.33 mg/kg dry weight) concentrations were identified in the upper bark mulch layers compared with concentrations in these substrates before they underwent the treatment experiments (i.e. Cu- below detection limit (<4 mg/kg dry weight), Zn- 20.5 mg/kg dry weight and Pb-6.3 mg/kg dry weight). Similar trends were observed for topsoil (Table 3-10). These data support the assumption that metals do not migrate far downwards in infiltration systems. Metal concentrations in the sand were less than both bark mulch and topsoil concentrations indicating that contaminants in this system may have migrated further downward, likely due to a combination of decreased filtration and adsorption from a greater hydraulic conductivity and absence of topsoil. Although a metal balance was performed on the substrate, net metal accumulation was greater than load applied, likely due to variation in contaminants within the substrate. Due to this variation in contaminants within the substrate, and limited samples collected, an accurate metal accumulation within each substrate was not possible.

Table 3-10- Substrate metal results compared with relevant ANZECC sediment quality guidance (low and high). Values are mg/kg dry weight. All samples are composite samples.

Contaminant		ISQG		Treatment System»	Sand		Sand/Topsoil		Topsoil	
(mg/kg dry weight)		Low	High	Substrate Analysed»	bark mulch	sand	bark mulch	topsoil	bark mulch	topsoil
Copper	Initial	65	270		<4	5	<4	6.8	<4	6.8
	Final				7.3	5.2	6.9	7.4	8.5	7.8
Zinc	Initial	200	410		20.5	31.0	20.5	49.3	20.5	49.3
	Final				48.0	33.0	53.0	48.0	62.0	51.0
Lead	Initial	50	220		6.3	6.8	6.3	15.5	6.3	15.5
	Final				4.7	6.4	9.3	15.1	8.0	15.4

3.1.7 Batch Leaching Investigations

Bench-top batch leaching experiments using stormwater (Section 2.1.3.2) were conducted to discern trends in pH, specific conductance and nitrate on individual substrate media as the mesocosm-experiments comprised mixed media. Results of the leaching experiments are shown in Table 3-11. The pH significantly decreased in the topsoil (pH 5.2) and bark mulch (pH 5.1) supernatant from pH in filtered tap water (pH 6.8), while sand beakers were elevated to pH of 7.1. Both bark mulch and topsoil beakers showed decreasing pH with extended time, likely because organic acids were slowly released in to the water. The pH in the sand system was not dependent on time, remaining at 7.1 throughout the beaker leaching investigation. The increased pH observed in the sand beaker is likely the result of the carbonaceous content of the unwashed sand. These data confirmed the direct effect of substrate type on effluent pH from leaching. Specific conductance was elevated above raw stormwater in all beakers, ranging from 185 (μS/cm) in the sand beaker to 462 (μS/cm) in the topsoil beaker. Specific conductance did not vary significantly with time. The experiments also confirmed nitrate was leaching from both the topsoil and bark mulch materials, evidenced by elevated contaminant concentrations as compared with untreated stormwater (Table 3-11). Nitrate was not detected above the 0.3 mg/L detection limit in the sand beakers, further confirming that sand was not a source of nitrate release. Reactive phosphorus was low or below detection limit in all three beakers.

Table 3-11- Batch leaching results of different substrates in stormwater. (Mean values) Time (5, 30, 60 min) shown from commencement of experiment.

	Beakers								
	Sand			Topsoil			Bark Mulch		
Time (mins)	5	30	60	5	30	60	5	30	60
pH (SU)	7.10	7.10	7.10	6.20	6.20	5.80	6.10	5.80	5.50
Specific Conductance (μS/cm)	185	192	189	351	462	385	200	251	263
Nitrate (mg/L)	<0.3	<0.3	<0.3	2.3	11.4	11.8	5.2	6.9	7.2
Reactive Phosphorus (mg/L)	<0.3	<0.3	<0.3	1.2	1.3	1.6	0.5	0.7	0.7

3.2 Summary

Elevated metal concentrations in untreated Christchurch stormwater far exceed the contextual ecotoxicological (ANZECC) guidelines recommended for healthy freshwater ecosystems.

Therefore, appropriate treatment should be implemented to mitigate adverse ecological impact prior to surface discharge. In line with the City Council's approach of implementing more 'natural' treatment systems, rain gardens are being adopted, but their effectiveness is not well understood as performance data is limited and not linked to design guidance.

This study found that total metal loads were reduced by 50% in all laboratory rain gardens of the design reported in this study, with the exception of copper in the topsoil-only system that demonstrated a negligible reduction due to its high dissolved fraction. However, nitrate was exported from the systems and so rain garden designs that incorporate topsoil will need to address this concern to avoid waterway impairment. Contrary to previous assumptions, topsoil is not an optimal substrate to enhance metal or nutrient removal in bioinfiltrative systems. This was attributed to its inability to buffer the pH of incoming stormwater resulting in higher dissolved metal fractions that are not conducive to being removed through settling like particulate metal phases.

Cu and Zn metal removal efficiencies were enhanced (Cu 56.4% and Zn 73.5%) when effluent pH was 7.38 (from sand systems) compared to an influent of pH 6.24 provided in raw stormwater. Therefore, pH enhancement afforded by alkaline substrates may help improve metal removal in bioinfiltrative systems, which are designed on settling principles. It may also provide sufficient pH enhancement in a topsoil-amended system to overcome the effects of a lower pH in topsoil, which is an important component for sustaining vegetation growth.

The experimental systems do not yet appear to have reached treatment capacity since metal removal efficiency was strongly correlated to contaminant loading and hydraulic conductivities did not decline following multiple treatment experiments. Furthermore, metal concentrations in the substrates were below the ANZECC ISQG substrate concentration thresholds.

Chapter 4: Mesocosm-Scale Alkalinity Supplementation Experiments

The primary water quality objective for stormwater management within New Zealand is to target removal of 75% of TSS (ARC 2003). It is assumed that other contaminants of concern such as particulate metals, particulate nutrients, and oil, grease and bacteria attached to sediments are concurrently removed (Smythe et al. 2007) (Section 1.4.2.1). However, dissolved metal concentrations are not necessarily removed with TSS and can impact downstream ecosystems. Dissolved metals are more bioavailable and thus more concerning than some of the particulate fractions, which are less toxic (Engstrom 2004). Laboratory experiments identified strong relationships between pH and dissolved fractions in heavy metals (Cu, Zn, and Pb) in stormwater employed in this research (Section 3.1.4.3) concurring with the literature (Dempsey et al. 1993; Landner and Reuther 2004). Therefore, mesocosm-scale rain gardens were established with a pH complement (mussel shells - a local waste product) to determine if elevated pH can increase contaminant removal efficiency of stormwater management systems.

4.1 Results and Discussion

4.1.1 Bench-top Batch Experiments

Bench-top leaching experiments (Section 2.1.3.2) were conducted to monitor a change in pH over 18 hours as a result of mussel shell amendment. These bench-top experiments effectively served as a proof of concept *a priori* the next mesocosm-scale experiments. Results from these tests indicated that untreated stormwater pH (6.36 ± 0.04) was substantially increased by adding mussel shells (Table 4-1). While the pH dropped to 5.19 in a beaker that contained only topsoil and stormwater (beaker 3), it increased up to pH 7.75 when 100 mL crushed mussel shells were added to stormwater (beaker 4). Organic acids (humic, fulvic, etc.) likely effected a reduction in the pH of stormwater, while carbonate in the mussel shells provided alkalinity (Cubillas et al. 2005; McCauley 2011). Beakers 10, 11 and 12, which contained a mixture of mussel shells and topsoil, recorded a drop in pH in the final measurement (18 hours), indicating that the organic acids are slowly released into solution. Although the pH was elevated during the first 60 minutes (beakers 10, 11, and 12) due to the mussel shell amendment, the subsequent drop in pH may be important in systems which are designed to

detain stormwater for extended periods of time. Generally, beakers with crushed mussel shells had a larger increase in pH compared to beakers with uncrushed shells.

Speciation of both Cu and Zn can vary widely between the observed pH range of pH 5.2 and 7.75, with particulate species more common at higher pH (Dempsey et al. 1993). Particulate phase metals are easier to remove through filtration systems than dissolved fractions (Fletcher et al. 2004; Pan and Duan 2011). Thus, if substrate additions can raise the influent pH and promote particulate metal speciation, enhanced metal removal (i.e. water quality treatment) is afforded.

Table 4-1- Mesocosm-scale pH complement batch leaching results. Beaker number and substrate are shown at the top of the table followed by pH (SU) at increasing time (1, 5, 15, 30, 60, 1080 min).

Material	**Beaker Number**											
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**
Stormwater (700 mL)	■	■	■	■	■	■	■	■	■	■	■	■
Soil (100mL)		■						■	■	■		
Soil (200 mL)			■								■	■
Mussel Shells (100 mL)				■				■				
Mussel Shells Crushed (100 mL)					■						■	
Mussel Shells (200 mL)						■			■			
Mussel Shells Crushed (200 mL)							■			■		■
Minutes	**pH Measurements**											
1	6.36	5.47	5.60	6.58	7.22	7.10	7.27	6.42	6.78	7.12	6.57	7.12
5	6.32	5.40	5.36	6.75	7.28	7.15	7.29	6.57	6.83	7.19	6.63	7.13
15	6.32	5.33	5.30	6.94	7.29	7.22	7.30	6.60	6.90	7.20	6.68	7.13
30	6.34	5.34	5.26	7.07	7.33	7.24	7.31	6.74	6.95	7.21	6.75	7.13
60	6.40	5.20	5.23	7.13	7.38	7.24	7.36	6.79	6.96	7.26	6.78	7.18
1080	6.42	5.23	5.19	7.46	7.75	7.49	6.54	6.99	7.10	6.52	6.42	6.61

4.1.2 Raw (Influent) Stormwater Characterisation

Stormwater employed in the mesocosm experiments reported in this chapter was collected from the same Addington catchment used for the first set of experiments reported in Chapter 3, and analytical results presented in Table 3-2 and Table 4-6 indicate it is of comparable quality. General water quality parameters for untreated stormwater collected from the stormwater feed tank are reported in Table 4-2. Stormwater in the feed tank included water collected on 8 July 2011 and 1 September 2011. Stormwater was collected on 22 February 2011, but was subsequently discarded for QA/QC purposes (lack of refrigeration after power cuts as a result of the 22 February 2011 earthquake). General water quality variables are reported in Table 4-2, while comprehensive metal and nitrate data (parameters of concern) are provided in Table 4-3.

Table 4-2- Water quality parameters for untreated stormwater in mesocosm-scale alkalinity supplementation experiments. Values represent mean ± standard deviation. Specific conductance and pH were continuously logged. (n= 10 for hardness, turbidity and TSS samples)

Parameter	Untreated Stormwater
pH (SU)	6.7 ± 0.1
Hardness (mg/L as $CaCO_3$)	19.9 ± 2.8
Specific Conductance (µS/cm)	95.9 ± 15.0
Turbidity (NTU)	95.8 ± 38.7
TSS (mg/L)	122.4 ± 28.7

The pH, hardness, specific conductance, turbidity and TSS of the (untreated) stormwater are consistent with literature on untreated stormwater quality (Pan and Duan 2011), which was previously discussed in Section 3.1.1.

All median total metal concentrations exceeded the 90% ANZECC trigger values, while these values were exceeded by a factor of 5.1 (Pb) to 12.6 (Cu), highlighting the degree to which 90% of species are not afforded protection from untreated stormwater contaminant concentrations (Table 4-3). All median dissolved metal concentrations, with the exception of Pb which is most common in the particulate state at stormwater pH (Morrison et al. 1990), exceeded 90% trigger values. First flush samples were collected in the field and are presented to show relative magnitude of first flush samples compared with homogenised header tank samples (mean and median). Exceedance factors were significantly larger for first flush metal

concentrations (16.9 for Cu, 12.3 for Zn, and 8.1 for Pb). These data support the position that urban stormwater needs to be treated prior to discharging to surface water bodies (discussed earlier in Section 1.1) to mitigate further ecological impairment. Nitrate concentrations in untreated stormwater are below 90% ANZECC guidance in both first flush and median concentrations. It is important to note that if rain garden effluent infiltrated to groundwater, it would only be subject to NZMOH drinking water standards, which are significantly elevated from the 90% species protection trigger value. Untreated stormwater did not exceed drinking water standards.

Table 4-3- Untreated stormwater concentrations of metals (total and dissolved) and nitrate in feed tank used for mesocosm-scale alkalinity supplementation experiments compared with the ANZECC (2000) surface water trigger values (90%) and NZDWS (2005). Values represent mean ± standard deviation and median concentrations (µg/L). First flush samples were collected in the first 20 minutes of storm events. (n = 4, 10 and 10 for first flush, mean and median values, respectively)

Contaminant		Untreated Stormwater (µg/L)			90% ANZECC Guidance (µg/L)	90% ANZECC Exceedance Factor			New Zealand Drinking-water Standards (2005) (µg/L)
		First Flush	Mean	Median		First Flush	Mean	Median	
Copper	-Total	30.4 ± 23.3	24.6 ± 4.9	22.7	1.8	16.9	13.6	12.6	2,000
	-Dissolved	8.2 ± 3.2	14.7 ± 4.1	16.5	-	4.6	8.2	9.1	-
Zinc	-Total	184.2 ± 95.1	168.1 ± 12.3	168.5	15.0	12.3	11.2	11.2	1,500
	-Dissolved	110.5 ± 80.0	99.0 ± 6.2	99.9	-	7.4	6.6	6.7	-
Lead	-Total	45.1 ± 15.3	31.7 ± 6.7	28.4	5.6	8.1	5.7	5.1	10
	-Dissolved	4.3 ± 2.6	1.3 ± 0.1	1.3	-	-	-	-	-
Nitrate		2,600.0 ± 510.0	752.0 ± 85.0	600.0	3,400.0	-	-	-	50,000

Notes: 1. The New Zealand Drinking-water Standards (2005) do not provide a standard for Zinc but give a 1,500 µg/L guidance value.

4.1.3 Hydraulic Performance

Hydraulic conductivity testing was conducted on mesocosm-scale laboratory rain gardens as outlined previously in section 3.1. Results of constant head hydraulic conductivity testing on mesocosm-scale systems with alkalinity supplementation are presented in Table 4-4. Initial and final saturated hydraulic conductivity are similar, indicating that systems did not clog throughout the experimental runs. The hydraulic conductivity increased with increasing mussel shell component in the system, which is attributed to the bulky nature of the shells (Table 4-4).

Table 4-4- Hydraulic parameters of mesocosm-scale alkalinity supplementation rain gardens prior to (initial) and following (final) contaminant removal efficiency experiments. Mean values are given ± standard deviation. Ranges are shown in parenthesis below. TS= topsoil system, MS= mussel shell complement

Hydraulic Conductivity (K_{sat})	**TS**	**TS/MS (4:1)**	**TS/MS (1:1)**
Initial (mm/hr)	279.8 ± 30.4 (232.4 - 269.7)	430.8 ± 14.1 (401.9 - 466.6)	703.9 ± 60.6 (589.3 - 818.7)
Final (mm/hr)	275.7 ± 23.4 (235.4 - 359.4)	426.7 ± 14.7 (396.8 - 466.6)	676.9 ± 28.9 (620.6 - 733.1)

Hydraulic conductivity values ranged from 279.8 mm/hr (initial) in the topsoil system (TS) to 703.9 mm/hr in the system with largest mussel shell amendment (TS/MS (1:1)). The topsoil system (TS) was consistent with silty sand, while systems with mussel shell amendment (TS/MS (4:1) and TS/MS (1:1)) were consistent with sandy gravels (Mays 2004). There was no significant change between initial (taken prior to removal efficiency experiments) and final (taken immediately after removal efficiency experiments) results. All three treatment systems were an order of magnitude larger than the minimum 13 mm/hr design guidance stipulated for rain garden designs (ARC 2003; MDE 2009; SFPUC 2009).

4.1.4 Real-time Water Quality Monitoring

Real-time water quality monitoring of pH, specific conductance and temperature (Section 2.3.2) was performed on effluent from each of the mesocosm-scale rain garden systems throughout experimental runs. Nitrate concentration was not logged due to demand for equipment by other researchers. Figure 4-1 and Figure 4-2 show trends in pH and specific conductance, respectively, over three replicated experimental runs from each of the three

systems (TS, TS/MS (4:1), and TS/MS (1:1)). Temperature was recorded to account for pH adjustment due to temperature change and to normalize electrical conductivity to 25°C (specific conductance). Average pH and specific conductance values are summarised in Table 4-5.

Table 4-5- Water quality in raw stormwater (inflow) and outflow (i.e. treated effluent) from all mesocosm-scale laboratory systems with alkalinity supplementation.

Parameter	Mean Inflow ± SD			Mean Effluent ± SD								
	Raw Stormwater			TS			TS/MS (4:1)			TS/MS (1:1)		
pH (SU)	6.66	±	0.11	6.68	±	0.08	6.86	±	0.13	7.73	±	0.39
Specific Conductance (µS/cm)	96	±	15	92	±	7	578	±	193	1523	±	756
n	900			277			243			251		

Distinct pH trends were identified throughout the experimental runs (Figure 4-1). Average effluent pH increased with increased amount of mussel shells in the system (e.g. pH 6.68, 6.86, and 7.73 in TS, TS/MS (4:1), TS/MS (1:1) systems, respectively). The pH in the TS/MS (1:1) system decreased throughout the experimental run, likely due to the slow release of organic acids from the topsoil within the system. Despite this decrease in pH late in the experimental run, effluent from TS/MS(1:1) containing equal volumes of topsoil and mussel shells increased the pH approximately one SU unit above influent stormwater (pH 6.66). The pH change within the system may actually be higher than was observed in the system effluent, due to the topsoil layer below the mussel shell amendment (refer to Figure 2-2B). The highest pH would likely be recorded at the bottom of the mussel shell layer, before draining through the topsoil which has been shown to reduce the pH (Section 4.1.1).

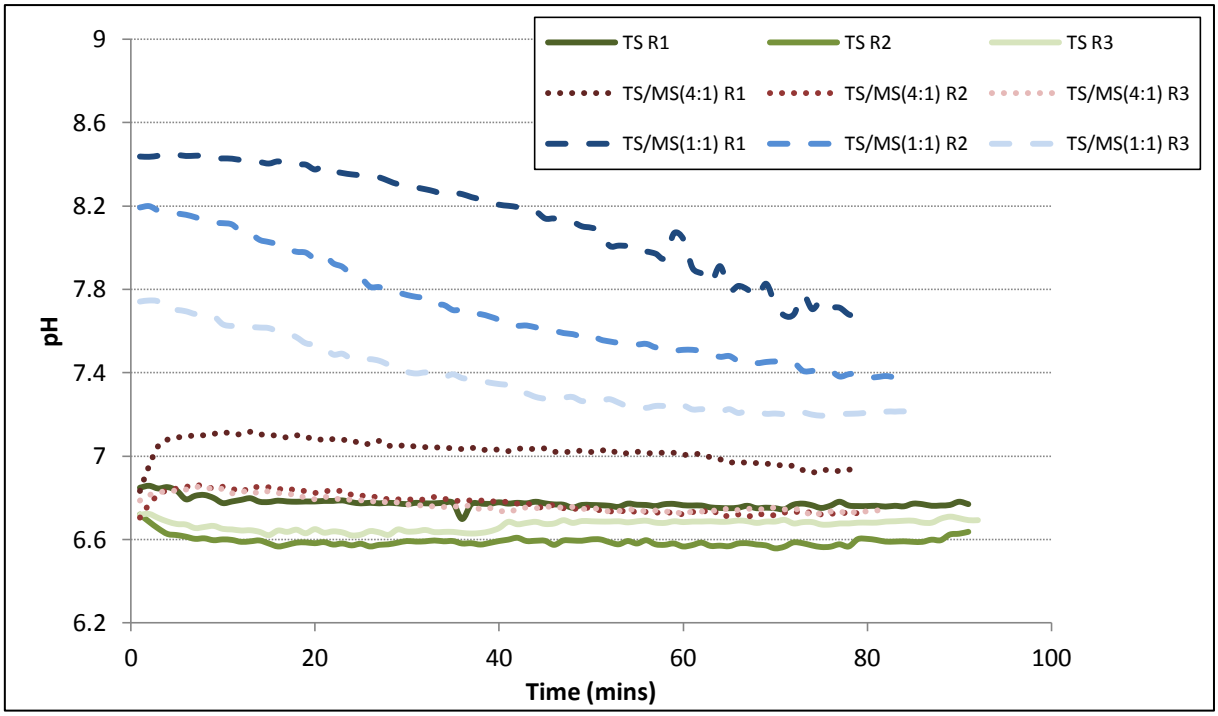

Figure 4-1- Continuous pH monitoring in mesocosm-scale alkalinity supplementation rain garden effluent throughout experimental runs.
Values were recorded every minute throughout experimental runs. Runs are shown separately and identified with (R1), (R2) and (R3) corresponding to run 1, run 2 and run 3, respectively.

Distinct trends were observed in specific conductance as a result of varying substrate. Of particular note were the systems with mussel shells (TS/MS(4:1) and TS/MS(1:1) that had specific conductance up to two orders of magnitude larger than topsoil-only systems (Figure 4-2). This increase likely resulted from carbonate release from the mussel shells, which is supported by the concurrent increased hardness observed for systems with mussel shells (Section 4.1.7). Specific conductance was reduced over the course of the experimental run in the TS/MS (1:1) system but remained almost a full order of magnitude larger than the topsoil only system (TS). A reduction in specific conductance with time may be explained by the reduction in carbonate dissolution as the stormwater drained through the system. Measurements for the TS/MS (1:1) system far exceed specific conductance documented for field performance of BMPs, which recorded average effluent at 209.44 ± 400.81 (Pan and Duan 2011), thus effluent from the systems would need to be compared with relevant discharge criteria stipulated in a consent.

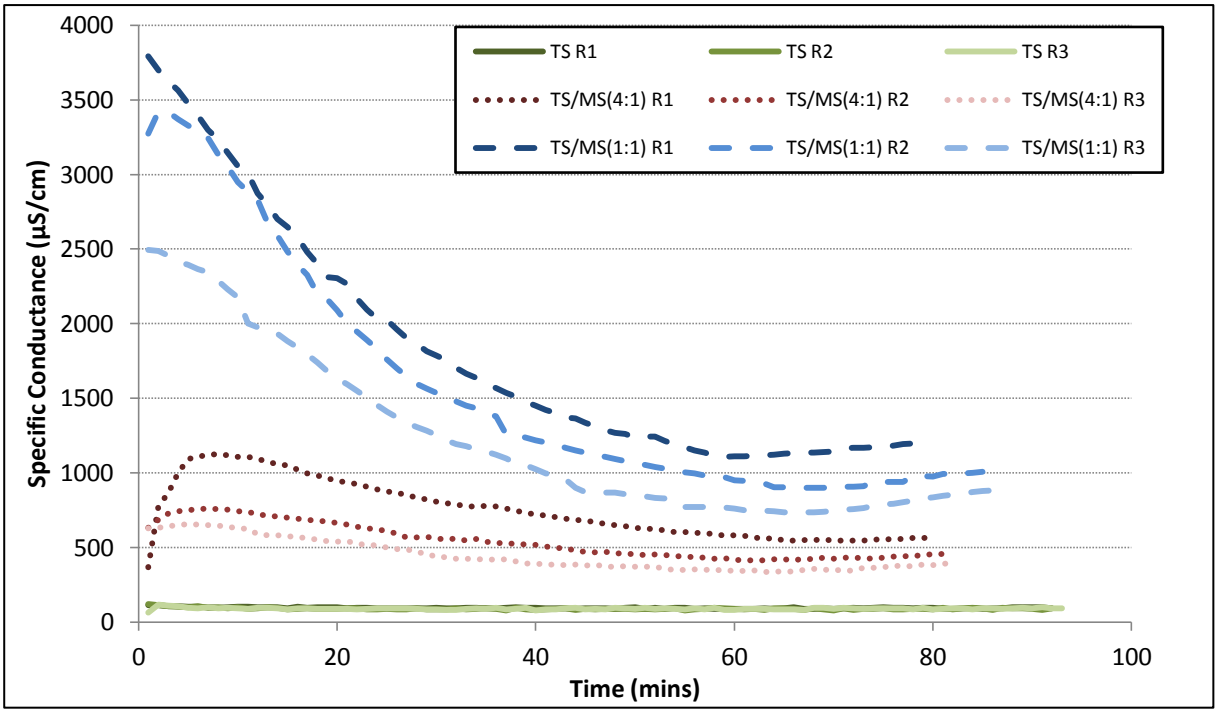

Figure 4-2- Continuous specific conductance monitoring in mesocosm-scale alkalinity supplementation rain garden effluent throughout experimental runs. Values were recorded every minute throughout experimental runs. Runs are shown separately and identified with (R1), (R2), and (R3) corresponding to run 1, run 2 and run 3, respectively.

4.1.5 Contaminant Removal Efficiency

4.1.5.1 Effluent Concentrations

All total mean and median metal (Cu, Zn, and Pb) were below their respective influent values showing contaminant removal within each system (Table 4-6). An exception was for dissolved Cu from the TS/MS (4:1) system, which had an average effluent concentration of 15.1 µg/L compared with the raw effluent of 14.7 µg/L. Average total metal concentrations for Cu, Pb, and Zn are lower in systems with mussel shells (TS/MS(4:1) and TS/MS(1:1)) compared with the TS system. High standard deviations for the TS system resulted from metal concentrations increasing as the experimental run progressed and effluent flow rate decreased (see Section 3.1.4.1), which was not observed for any systems containing mussel shells.

Table 4-6- Contaminant concentrations in outflow (i.e. treated effluent) from all mesocosm-scale alkalinity supplementation systems compared with untreated stormwater (inflow) and relevant 90% ANZECC guidance. Values represent median and mean ± standard deviation concentrations (µg/L). Percent dissolved metal fraction was calculated from mean concentrations. (n=10 for inflow and 24-28 for effluent)

Contaminant (µg/L)		**ANZECC Guidance (90%) (Unadjusted for hardness)**	**Inflow**				**Effluent**											
			Untreated Stormwater				**TS**				**TS/MS (4:1)**				**TS/MS (1:1)**			
			Median	Mean	±	SD	Median	Mean	±	SD	Median	Mean	±	SD	Median	Mean	±	SD
Cu	Total	1.8	22.7	24.6	±	4.9	16.7	24.5	±	23.5	20.3	21.83	±	5.7	15.3	15.8	±	3.5
	Dissolved		16.5	14.7	±	4.1	10.1	12.5	±	5.5	14.2	15.1	±	3.3	8.9	9.7	±	1.7
	% Dissolved			15%				51%				69%				61%		
Zn	Total	15.0	168.5	168.1	±	12.3	72.5	104.4	±	123.7	45.8	49.3	±	12.4	39.8	41.7	±	8.6
	Dissolved		99.9	99.0	±	6.2	41.9	55.4	±	52.5	22.9	23.5	±	5.7	17.1	17.4	±	2.9
	% Dissolved			59%				53%				48%				42%		
Pb	Total	5.6	28.4	31.7	±	6.7	6.8	8.6	±	4.5	5.4	6.5	±	3.8	4.1	4.2	±	1.1
	Dissolved		1.3	1.3	±	0.1	1.0	1.3	±	1.0	2.1	2.0	±	0.4	0.8	0.9	±	0.3
	% Dissolved			4%				15%				32%				21%		
Nitrate		3,400	600	752	±	85	3,800	4,600	±	1,452	2,900	3,400	±	1,100	2,700	2,600	±	742

Concentrations of total and dissolved metals from effluent in all systems were compared with influent concentrations and the (unadjusted) ecotoxicological ANZECC values (ANZECC 2000). Similar to untreated stormwater, all median total metal concentrations, with the exception of lead in the TS/MS (4:1) and TS/MS (1:1) systems, exceeded the unadjusted 90% species protection ANZECC trigger values (Table 4-6). There was little difference in Cu concentration at the influent (24.6 μg/L) and effluent from TS and TS/MS (4:1) (24.5 μg/L and 21.8 μg/L, respectively). The dissolved metal fraction was greater in TS/MS (4:1) and TS/MS (1:1) at 69% and 61% respectively compared with effluent from the TS system (51%). This increase in dissolved fraction may be in part due to Cu-carbonate complexes as a result of the mussel shell complement. These Cu-carbonate complexes can reduce the metal toxicity by reducing free metal ions in solution (Hyne et al. 2005). All dissolved Cu and Zn median concentrations exceeded the 90% guidance. However, due to the hardness measured in the TS/MS (4:1) and TS/MS (1:1) effluent, ANZECC values were modified for these systems. Hardness values for all three systems are shown in Table 4-7. The calcium carbonate from the mussel shells added alkalinity and increased the hardness in the system effluent. Increased hardness results in decreased metal toxicity and higher hardness modified metal trigger values (ANZECC 2000) in water with hardness above 30 mg/L as $CaCO_3$.

Table 4-7- Water hardness in influent and effluent from mesocosm-scale alkalinity supplementation experiments. Values represent mean ± standard deviation. (n=10 for influent and n=24-28 for effluent)

	Influent	**TS**	**TS/MS(4:1)**	**TS/MS(1:1)**
Hardness (mg/L as $CaCO_3$)	19.9 ± 2.8	22.4 ± 4.5	62.3 ± 12.5	101.7 ± 15.5

Effluent water hardness from the TS system was below 30 mg/L as $CaCO_3$, and thus TS effluent was compared to unadjusted ANZECC trigger values. ANZECC hardness modified trigger value (HMTV) algorithms used to calculate the revised trigger values for effluent in TS/MS (4:1) and TS/MS (1:1) are shown in Table 4-8.

Table 4-8- ANZECC hardness modified trigger value algorithms. (modified from ANZECC 2000)

Copper	$HMTV=TV(H/30)^{0.85}$
Zinc	$HMTV=TV(H/30)^{0.85}$
Pb	$HMTV=TV(H/30)^{1.27}$

Based on an average hardness of 62.3 mg/L as $CaCO_3$ and 101.7 mg/L as $CaCO_3$ (Section 4.1.7) in effluent from TS/MS (4:1) and TS/MS (1:1), respectively, new trigger values were established (Table 4-9). New hardness modified trigger values (HMTV) for TS/MS (4:1) were: Cu = 3.35 µg/L, Zn = 27.92 µg/L and Pb = 14.17 µg/L. Trigger values for TS/MS (1:1) were adjusted to Cu = 5.1 µg/L, Zn = 42.3 and Pb = 26.4 µg/L. The hardness in effluent from both system containing mussel shells made substantial differences to the potential ecotoxicity of discharged water. Cu (total and dissolved) still exceed the respective hardness modified ANZECC criteria; however, the exceedance factors are lower at 6.1 and 3.0 for TS/MS (4:1) and TS/MS (1:1) respectively, compared with 9.3 for topsoil only system (Table 4-9). Zn levels in the TS/MS (1:1) effluent did not exceed the hardness-modified ANZECC criteria of 5.1 µg/L.

Table 4-9- Contaminant exceedance factors of relevant ANZECC guidance in untreated stormwater (inflow) and outflow (i.e. treated effluent) from all mesocosm-scale alkalinity supplementation experimental systems. Exceedance factors are calculated from median concentrations (See Table 3-5). (n=10 for inflow and 24-28 for effluent) TS exceedance factors were calculated for ANZECC guidance 90%, while TS/MS (4:1) and TS/MS (1:1) were calculated using ANZECC hardness modified trigger values (HMTV). NA indicates no exceedance of applicable guidance.

Contaminant		ANZECC Guidance (90%) µg/L	ANZECC HMTV		ANZECC Exceedance Factors			
			TS/MS (4:1)	TS/MS (1:1)	Inflow	Effluent		
			µg/L	µg/L	Raw Stormwater	TS	TS/MS (4:1)	TS/MS (1:1)
Copper	Total	1.8	3.4	5.1	12.6	9.3	6.1	3.0
	Dissolved				9.2	5.6	4.2	1.7
Zinc	Total	15.0	27.9	42.3	11.2	4.8	1.6	NA
	Dissolved				6.7	2.8	NA	NA
Lead	Total	5.6	14.2	26.4	5.1	1.2	NA	NA
	Dissolved				NA	NA	NA	NA
Nitrate		3,400	-	-	1.1	NA	NA	NA

4.1.5.2 Contaminant Load

Cumulative contaminant loads were computed from flow-rate measurements and concurrent contaminant concentrations throughout experimental runs (Figure 4-3).

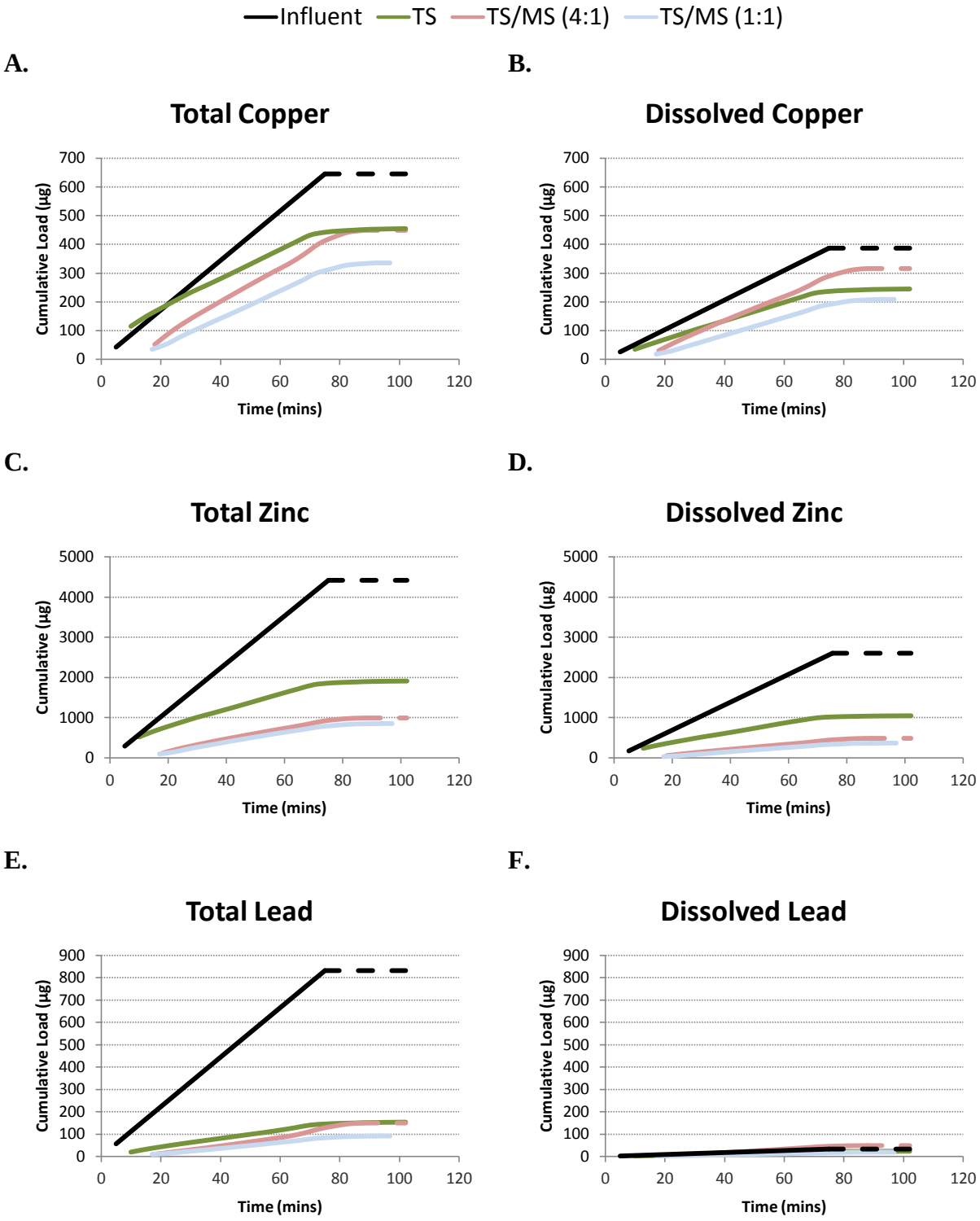

Figure 4-3- Cumulative total and dissolved influent and effluent metal load over time in mesocosm-scale alkalinity supplementation systems. Dashed lines indicate stormwater was no longer being applied/discharged from system and thus cumulative contaminant load remains constant.

All systems released less cumulative *total* metal load than was applied (Figure 4-3A, C, E), confirming removal of metal contaminants in each system. High total metal removal efficiencies for Pb and Zn were demonstrated from the difference between their respective influent and effluent amounts (Figure 4-3). Cumulative dissolved metal load was lowest in the TS/MS (1:1) system for Cu, Zn, and Pb, highlighting the importance of mussel shells in removal metals in these systems (Figure 4-3A, C, E). This is likely in part due to conversion of dissolved phase metal to particulate as a result of the elevated pH (pH 7.73). Dissolved fraction of Cu and Zn were much greater than Pb, which is commonly in particulate state in the observed pH range (Figure 4-3B, D, F) (Morrison et al. 1990).

Metal removal as a function of metal speciation was further examined by plotting the relative proportions of particulate and dissolved fractions that were discharged from each system (Figure 4-4). Removal efficiencies (%) were calculated based on contaminant load differences between influent and effluent.

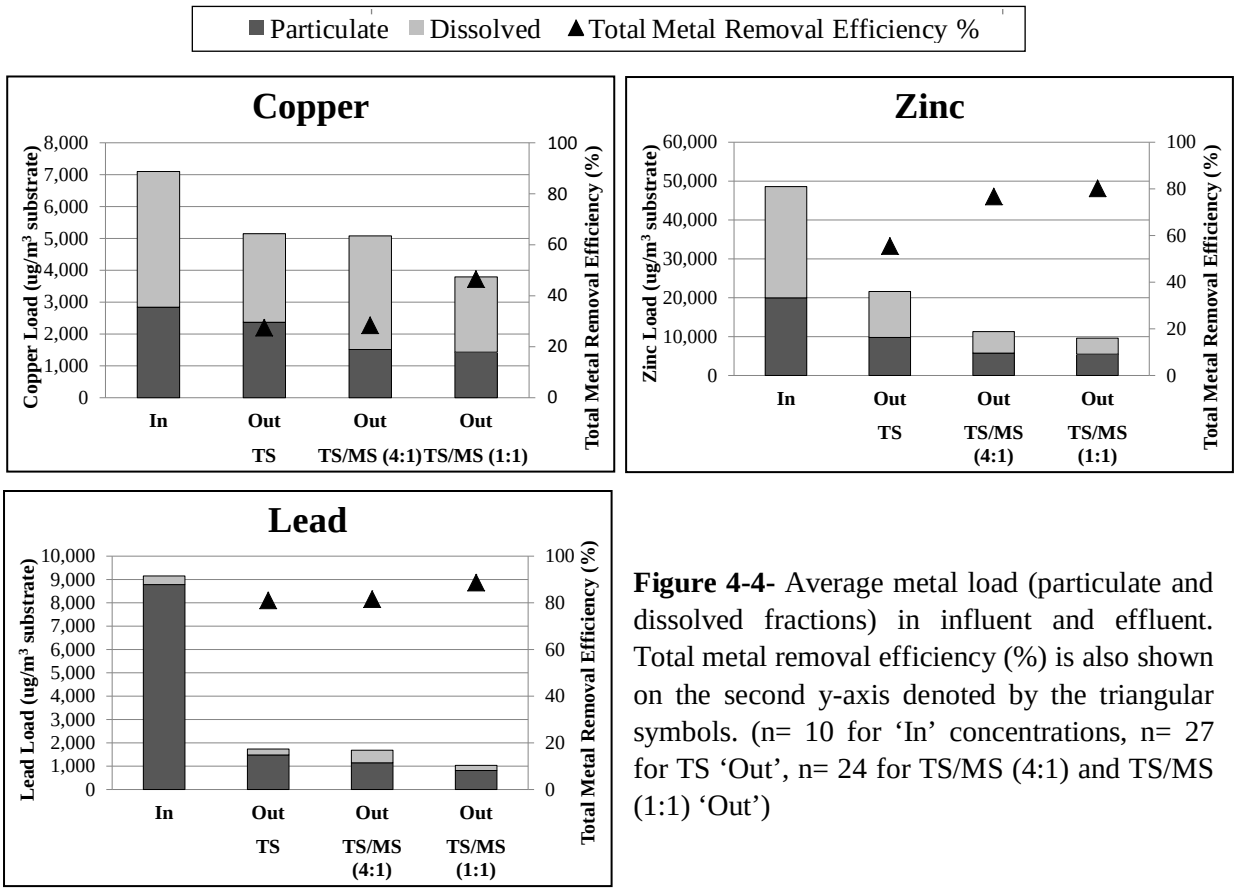

Figure 4-4- Average metal load (particulate and dissolved fractions) in influent and effluent. Total metal removal efficiency (%) is also shown on the second y-axis denoted by the triangular symbols. (n= 10 for 'In' concentrations, n= 27 for TS 'Out', n= 24 for TS/MS (4:1) and TS/MS (1:1) 'Out')

Metal removal efficiency increased with increasing mussel shell volume (Figure 4-4). Cu removal efficiency increased from 27% in the TS system to 47% in the TS/MS (1:1) system, while Zn removal efficiency increased from 55% in the TS system to 80% in the TS/MS (1:1) system (Figure 4-4). This was attributed to the alkalinity produced from the mussel shells that resulted in a higher pH discussed previously (Section 4.1.1). Removal efficiencies for the TS/MS (1:1) system were comparable with those removal efficiencies observed in the sand only system (Cu 56% and Zn 74%) (Chapter 3).

Dissolved copper in the effluent from the TS/MS (1:1) (61%) did not reduce as dramatically as in sand only system (24%) reported in Chapter 3. It has, however, been shown that water-soluble Cu is not an appropriate measure for bioavailable Cu as most of it occurs as non-bioavailable organic complexes (i.e. toxicity is correlated with the concentration of free metal ions rather than total dissolved metal concentration (Depledge et al. 2009). It is likely that only a portion of the metal was converted to particulate state and filtered out, as metal speciation shifts towards particulate phase above approximately pH 7 (Dempsey et al. 1993). The sand only systems in Chapter 3 raised the effluent pH to 7.38 and did not have significant organic content (i.e. no topsoil) to form dissolved organic complexes, thus Cu speciation was largely in the particulate fraction. The TS/MS (1:1) afforded higher pH in effluent (7.73), but contained more organic content (i.e. topsoil) and also increased carbonate amendment (mussel shells) which form dissolved complexes, thus speciation was not as weighted towards the particulate fraction as was observed in the sand only system in Chapter 3. While these dissolved fractions would be accounted for in the total loading calculations, resulting in reduced removal efficiency, they are less toxic than uncomplexed forms (Hyne et al. 2005). Although there is still significant dissolved fraction, especially in copper, this dissolved copper is likely to be less toxic to the receiving environment (ANZECC 2000). The mussel shells do not only raise the pH, but also reduce the toxicity of the metals due to a combination of increased copper carbonate complexes reducing free cupric ions and increased number of calcium and magnesium cations competing with the metal ions for complexing sites (Hyne et al. 2005; Minton 2005).

4.1.6 Nutrient Export

Median nitrate concentrations from the topsoil-only (TS) system (4.6 mg/L) exceeded the ANZECC guidelines (3.4 mg/L) and the median influent concentration (0.6 mg/L), indicating

net export of nitrate from the system (Table 4-10). This was most likely due to release from organic topsoil and bark mulch substrate. Additional discussion of nitrate export can be found in Section 3.1.3. Nitrate levels should be further investigated in systems with vegetation to more accurately access nutrient loads.

Table 4-10- Median nitrate concentration in mesocosm-scale alkalinity supplementation systems effluent compared to 90% ANZECC guidance. (n= 10 for inflow concentrations, n= 27 for TS outflow, n= 24 for TS/MS (4:1) and TS/MS (1:1) outflow)

Median Values (mg/L)	90% ANZECC Guidance	Inflow	Outflow		
		Stormwater	TS	TS/MS (4:1)	TS/MS (1:1)
Nitrate	3.4	0.6	4.6	3.4	2.6

4.1.7 Additional Water Quality Parameters

Additional water quality parameters (turbidity and TSS) were measured in effluent from each system as compared with influent stormwater (Table 4-11). Turbidity and TSS concentrations decrease slightly with increasing mussel shell substrate but are only slightly below influent values. Turbidity and TSS in the TS system effluent were above influent levels. The marginal increase in both turbidity and TSS (95.8 NTU to 121.1 NTU for turbidity and 122.4 to 145.0 mg/L for TSS) is consistent with results in Chapter 3 for the topsoil system. Turbidity in effluent for all systems (TS, TS/MS (4:1), and TS/MS (1:1)) was below levels observed in sand systems (349 NTU) in Chapter 3. Nonetheless, elevated TSS and turbidity levels can impact negatively on stream environments and should be monitored for a longer period of time to determine if the elevated readings are due to the "newness" of the systems. TSS removal efficiency in biofiltration systems vary widely, ranging from a 3% average increase reported by (Pan and Duan 2011) to 97% removal (Fletcher et al. 2004).

Table 4-11- Average water quality parameters in mesocosm-scale alkalinity supplementation system effluent. (n= 10 for influent concentrations, n= 27 for TS effluent, n= 24 for TS/MS (4:1) and TS/MS (1:1) effluent)

Parameter	Influent	Effluent		
	Stormwater	TS	TS/MS (4:1)	TS/MS (1:1)
Turbidity (NTU)	95.8 ± 38.7	121.1 ± 21.4	85.6 ± 17.5	67.3 ± 7.9
TSS (mg/L)	122.4 ± 28.7	145.0 ± 23.7	112.3 ± 24.3	105.6 ± 12.8

4.2 Summary

Although incorporating the waste product mussel shells into rain garden designs (at a mesocosm-scale) offers a promising outlook for enhancing the metal removal efficiency in stormwater treatment systems, effluent contaminant concentrations did not typically meet the contextual 90% ANZECC guidelines. Nonetheless, effluent concentrations from systems with mussel shell amendments were closer to their respective hardness modified guidelines compared with the TS system, which was compared with the unadjusted 90% trigger values. Overall, comparing effluent concentrations with ANZECC trigger values is a conservative approach, as effluent concentrations are higher than they would be at the edge of the mixing zone where concentration compliance is stipulated. Additionally, numeric effluent criteria are rare for stormwater regulations in an international context, with the exception in the USA where total maximum daily loads (TMDLs) are sometimes in place.

Increased metal removal observed in systems containing mussel shells was attributed to the processes of enhanced adsorption and filtration facilitated by higher alkalinity and an elevated pH compared with the influent. Implications of this in stormwater treatment designs is that elevating effluent stormwater pH through choice of substrates can result in improved metal treatment capacity for systems that rely on biofiltration. Shells have been used in passive treatment systems to treat metal contaminated mine drainage (Daubert and Brennan 2007; McCauley et al. 2009); however mussel shells use has not been extended to full scale stormwater treatment to date.

Chapter 5: Field-scale “Rain Garden” Experiments

A field-scale rain garden was monitored across wet seasons to determine contaminant removal capacity and hydraulic performance (Section 2.2). Stormwater was collected from six storm events between 16 July 2010 and 1 September 2011 for use in laboratory experiments (Section 2.1). Additionally, flow was monitored through rain garden #12 (Figure 2-10) from 22 June 2011 (winter) through 18 September 2011 (spring).

5.1 Results and Discussion

5.1.1 Storm Characteristics

Storms were sampled throughout the year across different seasons. Stormwater employed in this research was collected from six targeted rain events based on a range of antecedent dry period (3-12 days), rainfall intensity (2.33-6.65 mm/hr), appreciable runoff duration (125-567 minutes) and total volume (1.67-25.36 mm) of precipitation as summarised in Table 5-1.

Table 5-1- Storm monitoring event characteristics including duration, intensity, total precipitation and antecedent dry period. All storm weather data was recorded at the UC Environmental Engineering Laboratory weather station with the exception of storm 6, which used NIWA monitoring station data due to a power cut to the UC weather station.

Event #		**1**	**2**	**3**	**4**	**5**	**6**
Date		**16-17 July 2010**	**3 Sept. 2010**	**5 Nov. 2010**	**22 Feb. 2011**	**8 July 2011**	**1 Sept. 2011**
Storm Duration	(mins)	145	162	567	125	151	145
Max. Intensity	(mm/hr)	6.02	2.33	13.06	3.09	6.65	3.48
Total Rain	(mm)	6.05	2.25	25.36	1.67	3.94	1.76
Ant. Dry Period	(days)	6	3	6	8	8	12

High initial and sustained rainfall intensity can increase runoff volumes such that runoff overwhelms stormwater detention and mitigation systems. Stormwater treatment systems are designed with high flow stormwater bypass, to safely convey water that is in excess of treatment system capacity. This ensures that systems do not backup and flood neighbouring areas in large storm events, but facilitates discharge of untreated stormwater. Flow monitoring in the Paul Matthews and Waitakere Vehicle Testing rain gardens in Auckland document stormwater bypass of rain gardens in seven out of the nine reported storm events (Trowsdale and Simcock 2008; Jayaratne et al. 2010). The Paul Matthews rain garden was reported to

have flow bypass in storm events with as little as 4.7 mm/hr rainfall intensity (6 minute average) (Trowsdale and Simcock 2008). Volume of runoff is as important as rainfall intensity, as it the time of peak intensity relative to the start of the storm. Even a relatively low peak can result in bypass if it occurs at the tail end of a long duration event. Rainfall intensity over time in Addington systems is shown for all monitored storm events in Figure 5-1. Highest rainfall intensity was recorded up to 13.06 mm/hr in the 5 November 2010 storm event; however, no rain garden bypass was observed. Annual rainfall in Christchurch is approximately half that of Auckland (NIWA 2010), thus stormwater is less likely to bypass similarly sized treatment systems in Christchurch. Average annual precipitation data for Christchurch and Auckland is included in Appendix E.

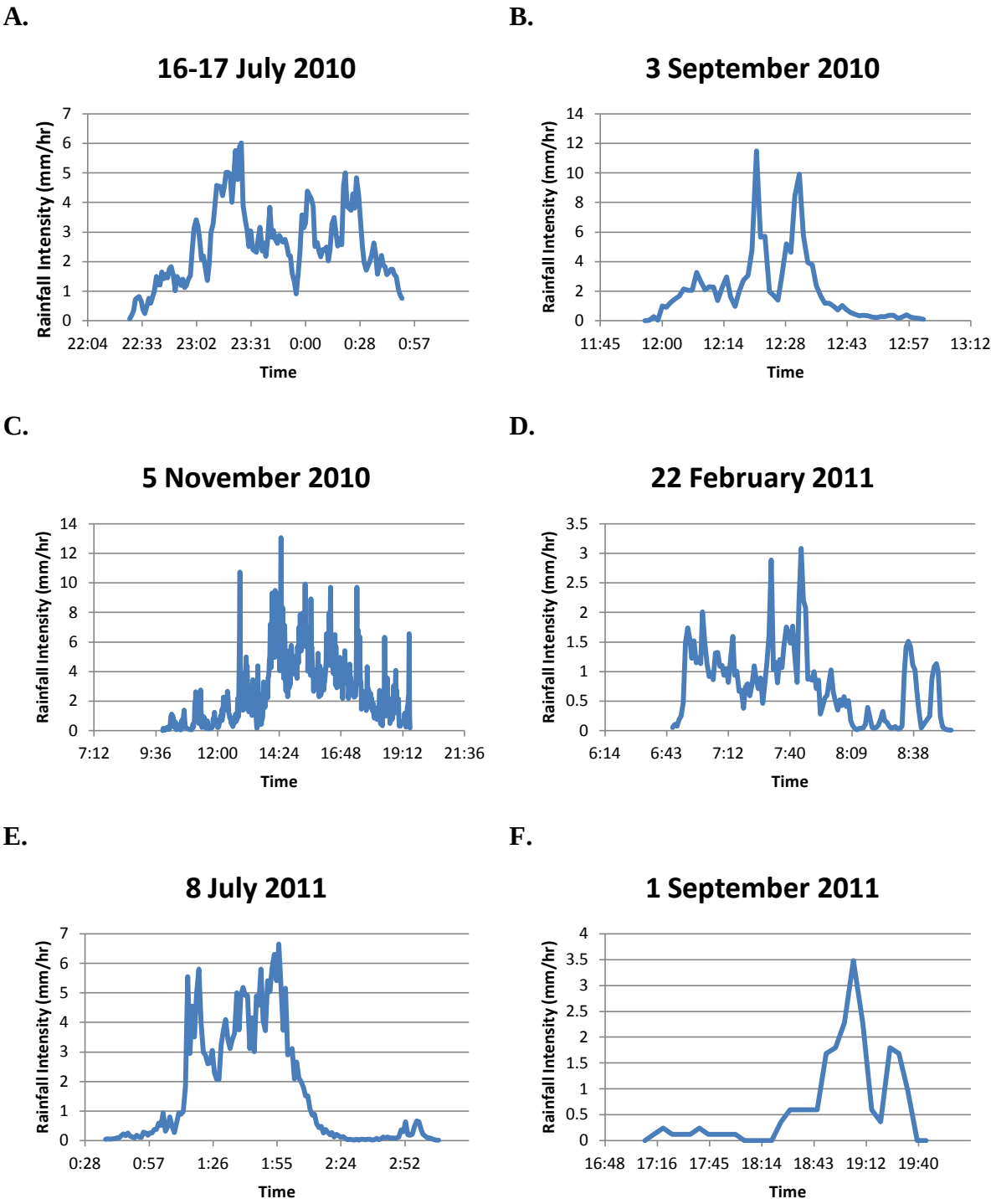

Figure 5-1- Rain intensity for individual storm events throughout field rain garden monitoring. Rain intensity was recorded at the UC Environmental Engineering Laboratory weather station using a laser disdrometer.

5.1.2 Contaminant Concentration

5.1.2.1 Effluent Concentrations

Concentrations of total and dissolved metals from effluent of operational field scale rain garden #12 (Section 2.2.2) were compared with their respective influent concentrations and the relevant ecotoxicological ANZECC values (ANZECC 2000) (Table 5-2). The data presented in Table 5-2 shows influent (and effluent) concentrations for events 1, 3, 5, and 6. This data represents samples collected in the field while influent data reported in Section 3.1.1 and Section 4.1.2 represents samples collected from the stormwater header tank prior to use in experimental runs. Field samples collected from event 2 (3 September 2010) and 4 (22 February 2011) were not analysed due to major earthquakes in Canterbury on 4 September 2010 and 22 February 2011, resulting in power loss and closure of the University. EMC calculations were not possible due to influent flow estimation constraints discussed in Section 2.1.4.1 and Section 5.1.3. Effluent sampling was limited by low outflow from the rain garden further discussed in Section 5.1.3.

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Table 5-2- Contaminant concentrations in inflow (untreated stormwater) and outflow for field rain garden system. Values represent median and mean ± standard deviation concentrations (µg/L). (n= 28 for inflow and 12 for outflow) Percent dissolved metal fraction and removal efficiency were calculated from mean concentrations. The 90% ANZECC guidance and NZMOH Drinking-water Standards (2005) are shown for comparison.

Contaminant (µg/L)		ANZECC Guidance (90%)	Inflow Untreated Stormwater			Outflow Discharged to Surface Water			Removal Efficiency	NZMOH Drinking-Water Standards (2005)
			Median	Mean	± SD	Median	Mean	± SD		
Copper	Total	1.8	12.8	16.6	± 14.0	15.9	15.6	± 2.4	6.0%	2,000
	Dissolved		3.6	4.7	± 2.8	12.8	12.2	± 2.4	-161.6%	-
	% Dissolved			25%			78%			
Zinc	Total	15.0	211.3	159.3	± 107.1	139.6	160.0	± 49.7	-0.5%	1,500
	Dissolved		73.3	77.6	± 64.4	115.4	128.2	± 48.3	-65.2%	-
	% Dissolved			49%			80%			
Lead	Total	5.6	26.0	38.5	± 28.3	11.7	11.4	± 4.9	70.4%	10
	Dissolved		2.1	2.5	± 2.4	1.0	1.0	± 0.5	59.9%	-
	% Dissolved			7%			9%			

All total mean and median metal concentrations in the influent and effluent of the field ‘rain garden’ system # 12 were far in exceedance of the ANZECC 90% species protection guidelines but well below the NZMOH drinking water standards (applicable for infiltration to groundwater) (Table 5-2). One exception was for total lead that had a median influent concentration of 26.0 µg/L and a median effluent concentration of 11.7 µg/L, which exceeded the drinking water standard of 10 µg/L. Mean Zn concentrations in stormwater inflow (159.3 µg/L) was far greater than mean concentration of Cu and Pb; however, Zn concentrations were consistent with the literature. For example, monitoring of influent to 84 stormwater systems identified average total zinc concentrations of 176.30 ± 132.33 µg/L (Pan and Duan 2011). The Zn is likely a result of leaching from galvanized roofs in the area and discharge from automobile components (brakes, tyres, body) (Section 1.1). Mean Cu and Pb concentrations in untreated stormwater (i.e. inflow) are below mean concentrations reported in the literature but well within the range of concentrations observed in stormwater (Pan and Duan 2011).

Dissolved Cu (78%) and Zn (80%) fractions in the effluent were much higher than dissolved influent proportions (25% Cu and 49% Zn) highlighting that the metal speciation of the influent stormwater changes substantially as it passes through these field systems, evidenced by negative removal efficiencies for dissolved Cu and Zn (Table 5-2). The effect is more pronounced for Cu and Zn and was only marginal for Pb, which is commonly in the particulate state at influent pH (Morrison et al. 1990). This has important implications for stormwater treatment given the enhanced ecotoxicity of dissolved metal fractions (Depledge et al. 2009). If the effluent solely discharges to groundwater, then this ecotoxicity effect is not of concern as concentrations do not exceed the NZMOH drinking water standards most applicable for systems discharging to groundwater. However, if the effluent is discharged to surface waters, elevated concentrations (above 90% ANZECC species protection levels) could impact negatively on the receiving ecosystem (Harding 2005).

These Addington field systems were designed and fit with an under-drain to convey discharge to surface water mainly in the event of ‘excess’ discharge that could not drain to groundwater (i.e. stormwater volume exceeds infiltration capacity due to poor permeability of soils in the area during high precipitation events). Actual volumes discharging to groundwater are discussed further in Section 5.1.3.

The total metal removal efficiency was good for lead (which tends to dominate in the particulate phase) at 70.4% but poor for Cu at 6.0% and Zn at -0.5%, which are known to be prevalent in dissolved states within the stormwater signature (Gnecco et al. 2008) (Table 5-2). Dissolved fractions are more difficult to remove than particulates through filtration systems (Fletcher et al. 2004). The range of concentrations in the influent during each of the four events were much higher than the effluent ranges (evidenced by their standard deviations), which is explained by the initial higher first flush concentrations in the influent at the onset of the storm runoff (Figure 5-2). Metal concentrations were higher in the initial 10 minutes of runoff compared with the remainder of the storm events (Figure 5-2) in line with elevated first flush trends reported in the literature (Mason et al. 1999; Pennington and Webster-Brown 2008; Wicke et al. 2010). A low initial storm intensity reported on 1 September 2011 (Section 5.1.1, Figure 5-1F) corresponded with a lower first flush concentration for Cu, Pb, and Zn (Figure 5-2). These data support literature findings that increased rainfall intensities result in elevated stormwater contaminant concentrations due to an increased capacity to mobilise contaminants (Egodawatta et al. 2009). Contaminant concentrations reached a relative steady state by the second or third sample corresponding to 10 and 20 minutes, respectively.

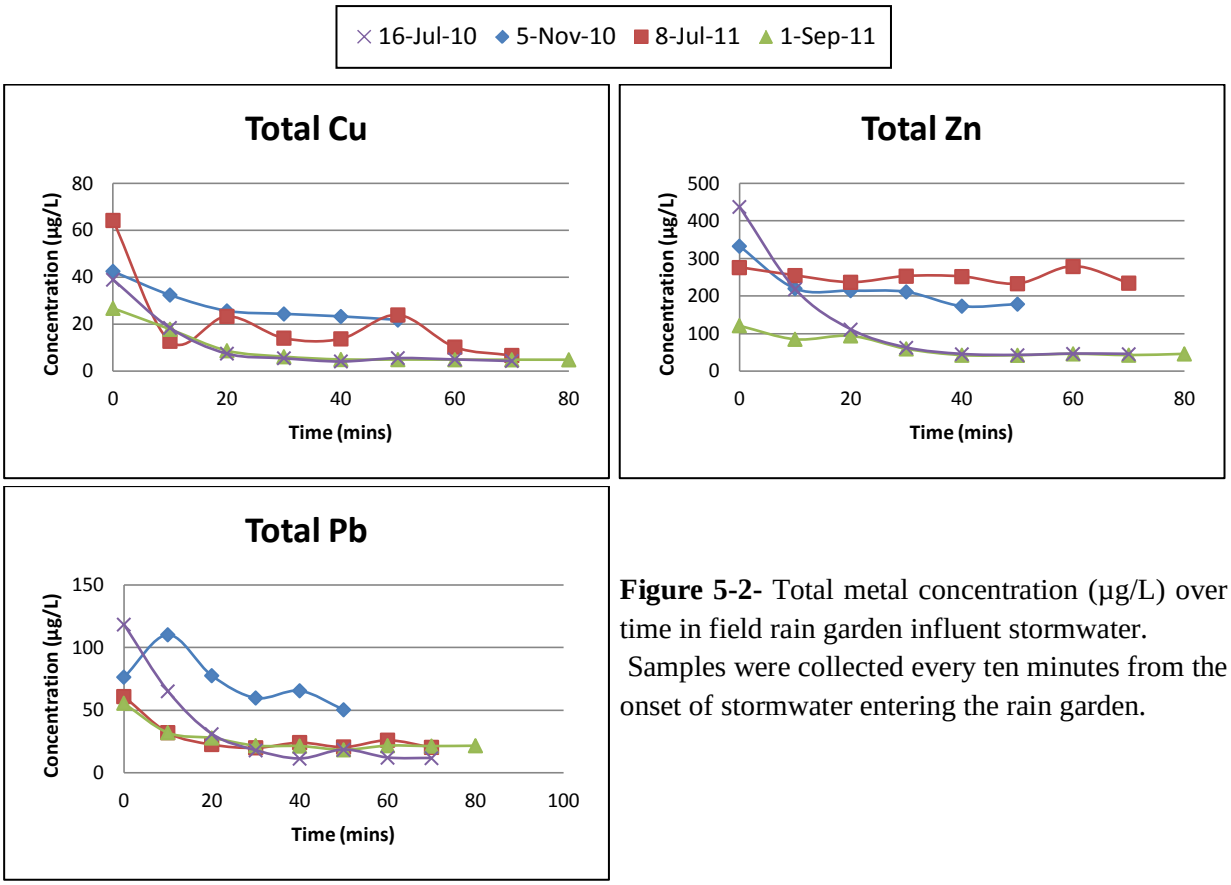

Figure 5-2- Total metal concentration (µg/L) over time in field rain garden influent stormwater. Samples were collected every ten minutes from the onset of stormwater entering the rain garden.

5.1.2.2 Additional Water Quality Data

Additional water quality parameters were measured in the influent and effluent of the field rain garden systems (Table 5-3). Despite low metal removal efficiencies (Zn and Cu) in the field systems, turbidity and TSS were reduced as stormwater passed through the field rain garden system. Turbidity and TSS were reduced by 85 and 91 %, respectively (Table 5-3). The current treatment water quality design criteria for rain gardens in New Zealand (Auckland) (Section 1.4.2.1) adopt the TP10 recommendation of 75% TSS removal (ARC, 2003), which was met in the monitoring period of this system. Nitrate reduction and slight increase in specific conductance (-43% and 11.5% change, respectively) were due to small variation at low concentration, and are not an accurate indication of change through the system. While laboratory mesocosm-scale systems employing AP-20 grade sand produced higher pH and hardness values in system effluent (Section 3.3.5), field rain gardens, designed with the same AP-20 sand, did not afford the same change to effluent water. The pH and hardness in field systems did not change substantially between influent and effluent concentrations (Table 5-3), indicating that the carbonate content in the unwashed laboratory systems may be depleted.

Table 5-3- Storm influent and effluent water quality parameters for field rain garden system. Values represent mean ± standard deviation with differences between the two given as a percentage increase or decrease. (n=25 for influent, n=12 for effluent)

Parameter	Influent			Effluent			% Change
pH (SU)	6.45	±	0.1	6.67	±	0.1	3%
Turbidity(NTU)	105	±	18	15.9	±	7.5	-85%
TSS (mg/L)	166	±	46	15.6	±	5.8	-91%
Specific Conductance (µS/cm)	58.3	±	5.7	65	±	4.3	2%
Nitrate (mg/L)	0.7	±	0.2	0.4	±	0.2	-43%
Hardness (mg/L as $CaCO_3$)	20.5	±	7.1	20.33	±	4.5	-1%

5.1.3 Flow Monitoring

It was intended to monitor flow throughput in the field systems for longer period but a combination of trouble-shooting and accurately calibrating field logging equipment, coupled with repeated disruptions arising from the multiple earthquakes, hindered the anticipated period of logging. Nonetheless, effluent flow was monitored during the wettest season from 22 June 2011 (winter) through 18 September 2011 (spring) using water capacitance data loggers that measured the height of water above a v-notch weir (Section 2.2.3).

Higher rain intensity generally corresponded to larger total daily precipitation marked by events A through E in Figure 5-3 and Figure 5-4. Results of the effluent monitoring indicated that very little water was actually discharged to surface water (Figure 5-5). Despite the range of rainfall events (Figure 5-3) during the wettest part of the year (NIWA 2010), outflow from the rain garden was minimal in the three months of monitoring and were very small in all cases (Figure 5-5). The largest flow recorded at the rain garden outlet was 9 L/hour on the night of 28 June 2011, which was only sustained for 15 minutes with total discharge from this event amounting to 5 L (Event A in Figure 5-3, Figure 5-4, and Figure 5-5). The total precipitation recorded for this storm event was 17.2 mm, amounting to approximately 16.1 m^3 of runoff entering the rain garden based on calculations using the SCS curve number method outlined in Section 2.2.3. The rain data (Figure 5-3and Figure 5-4) corresponds directly with small outflow from the rain garden (Figure 5-5) with the exception of event B, which may be due to weather variation between the weather station and the rain garden. No daily discharge was recorded greater than 5 litres in the three months of monitoring. In approximately three

months of monitoring only 24 L of water were discharged from the rain garden and greater than 99.5% of the runoff entering the system infiltrated to ground water.

5.2 Summary

The Addington rain garden operated extremely well at detaining and infiltrating stormwater during the monitoring period. Despite exceedances of ANZECC 90% guidelines in effluent samples, very little was discharged to surface water, and is thus unlikely to contribute to any significant environmental degradation. ANZECC guidance values are designed for ecological protection of surface water communities and thus are not applicable to groundwater infiltration. Although the Addington rain garden system was designed to convey stormwater to surface water (similar to TP-10 guidance), the majority of the stormwater was infiltrated, and thus ANZECC guidelines may not be the most appropriate benchmark. Additionally, the effluent water that was sampled from the rain garden outlet was only representative of what was discharged to surface water (<0.5%), although it is believed that infiltrated water would be of similar water quality. Cu and Zn were below NZMOH drinking water standards, while Pb only marginally exceeded the 10 μg/L level.

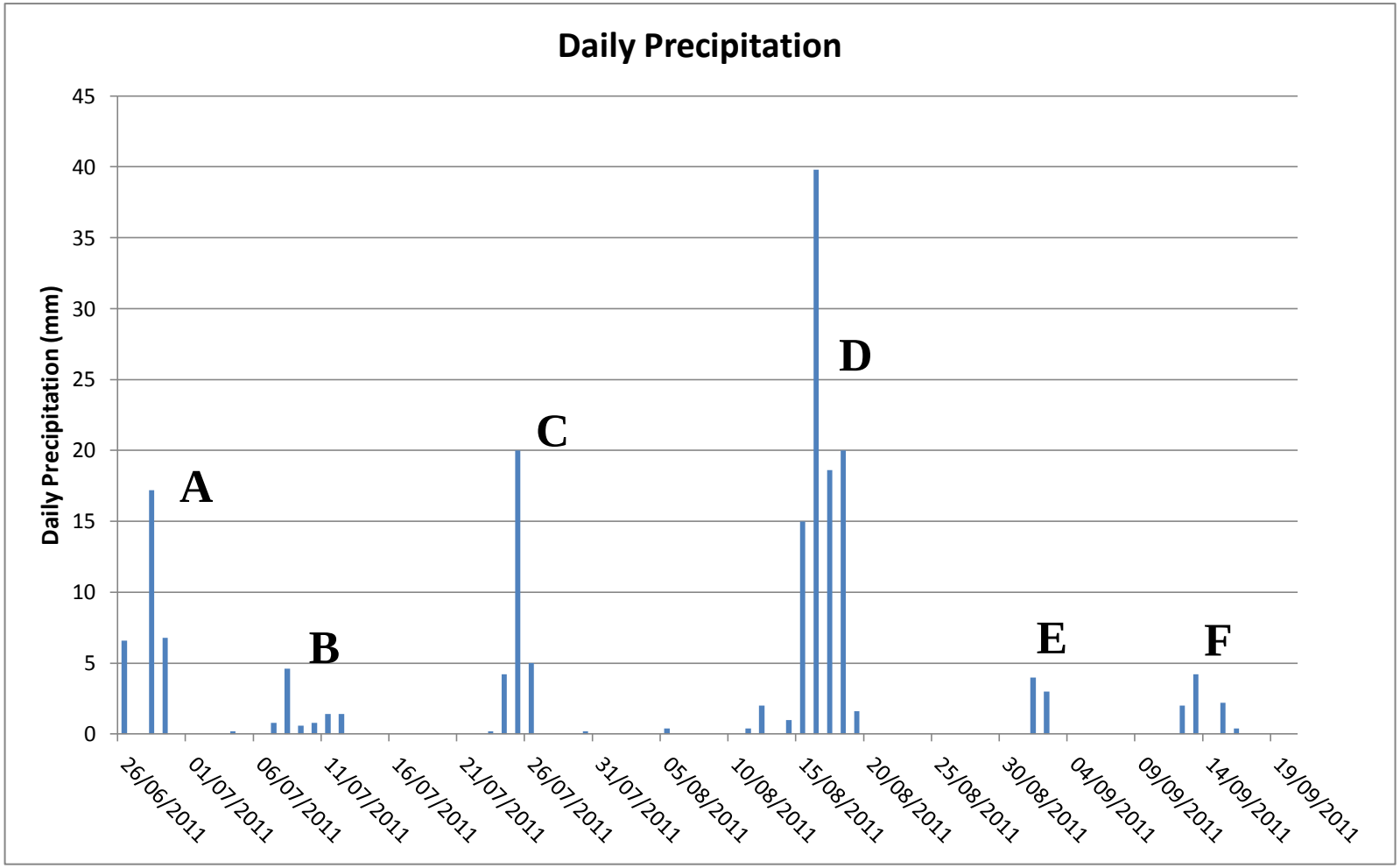

Figure 5-3- Daily precipitation data corresponding to field outlet flow monitoring period. Letters A-F represent appreciable storm events/series of storm events. Daily precipitation data was recorded at the UC Environmental Engineering Laboratory weather station using a 0.2 mL tipping bucket.

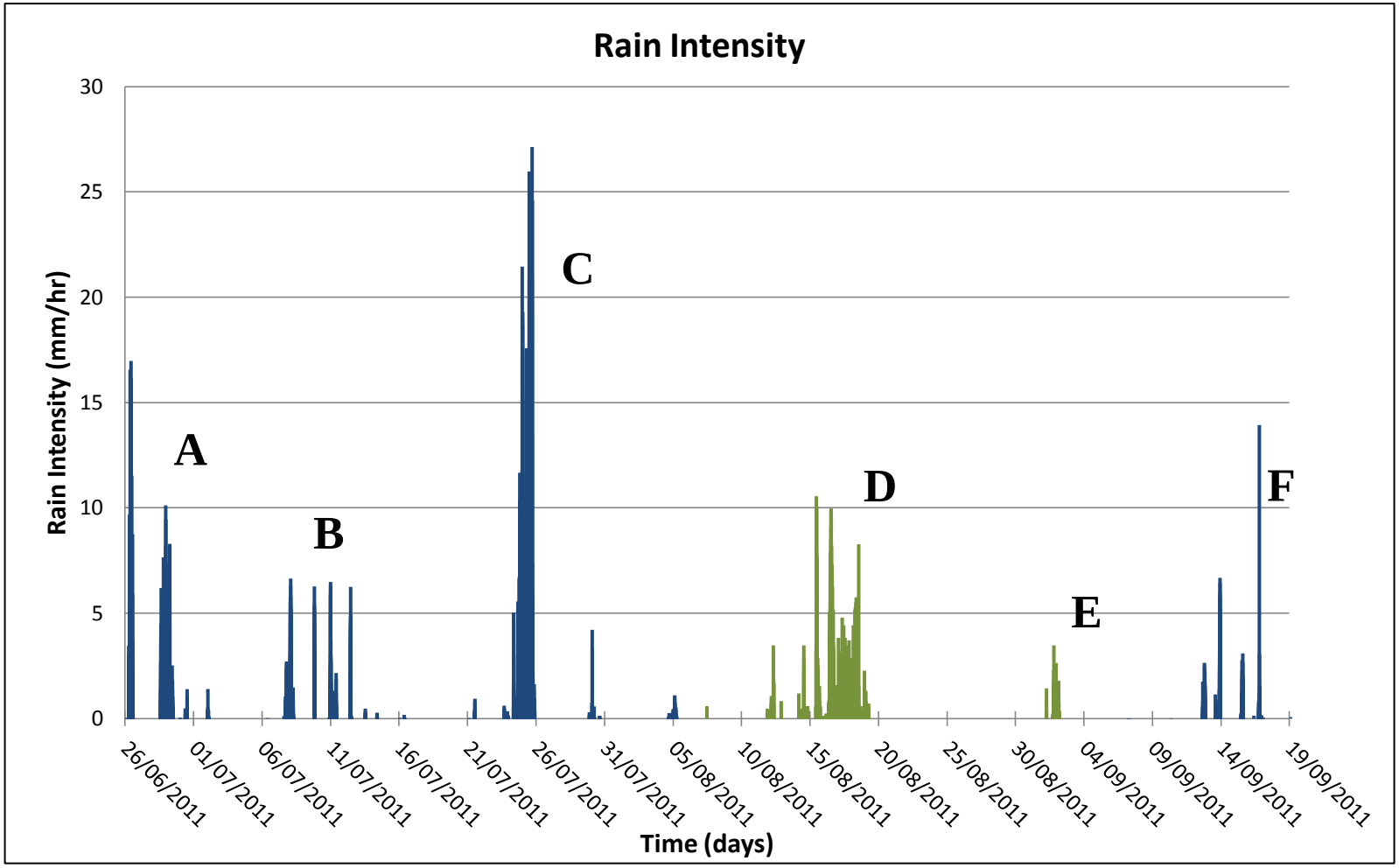

Figure 5-4- Rain intensity storm data corresponding to outlet flow monitoring period. Letters A-F represent appreciable storm events/series of storm events. Rain intensities (1 min. interval) were recorded at the UC Environmental Engineering Laboratory weather station using a laser disdrometer (blue series), except between 7 August 2011 to 2 September 2011. Rain intensity data for that time has been substituted with NIWA monitoring station data (5 min. interval, green series).

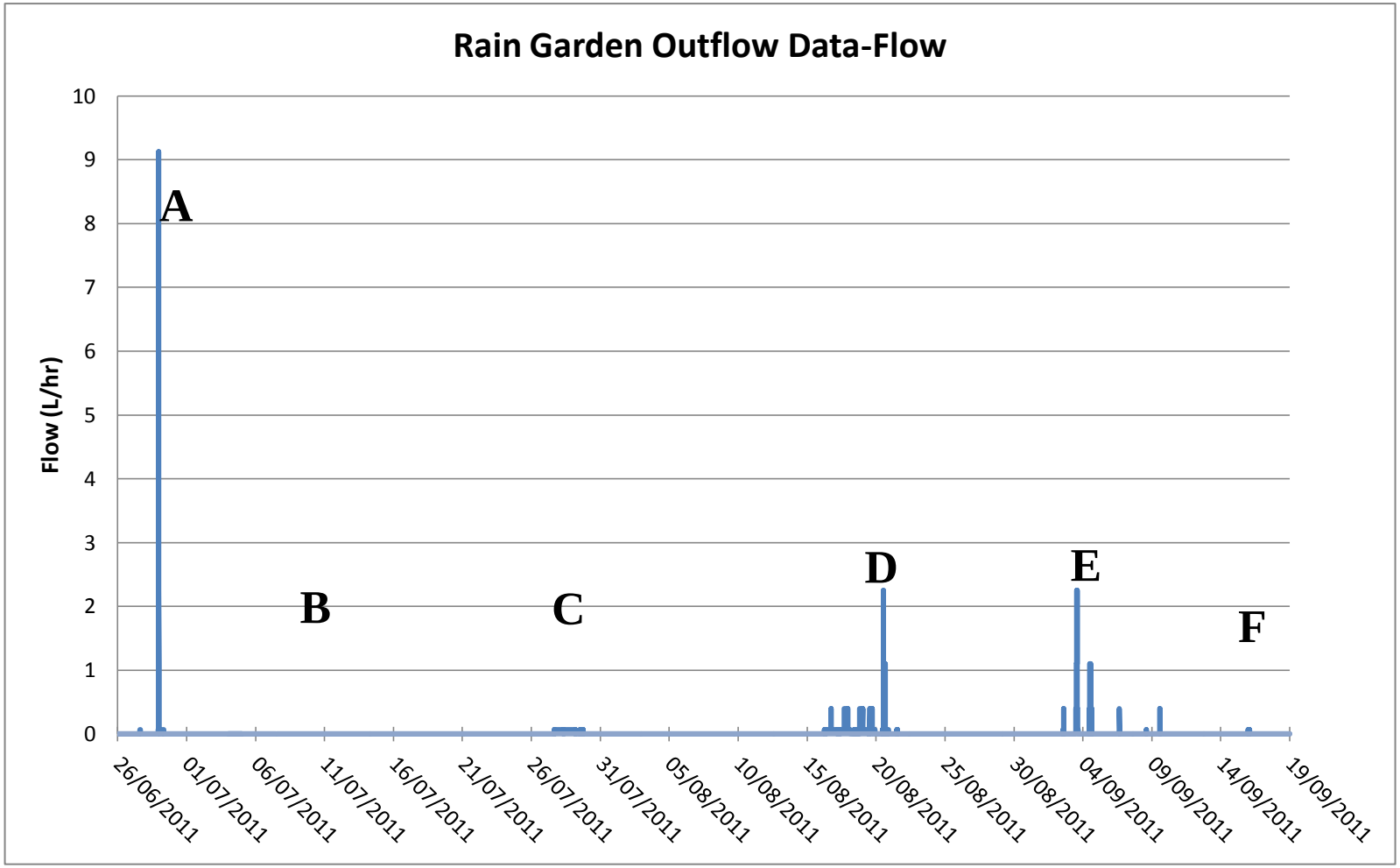

Figure 5-5- Rain garden outflow data discharged to surface water. Numbers 1-6 represent appreciable storm events/series of storm events.

Chapter 6: Conclusions and Research Recommendations

6.1 Research Summary

The first major objective of this research was to investigate effects of topsoil proportions and pH enhanced substrates on (a) hydraulic conductivity and (b) metal removal efficiency in mesocosm-scale laboratory rain gardens. Nationally, the treatment goals for rain gardens stipulated by governing consenting authorities are not well developed or consistent (Smythe et al. 2007), so developing a better understanding of their effectiveness based on design criteria could help inform robust stormwater design standards in New Zealand and overseas. Laboratory systems were successfully established and operated, the results from which contribute to the limited knowledge of biofiltration hydraulic and contaminant treatment (Fletcher et al. 2004; Dietz and Clausen 2005; Henderson et al. 2007)

The second major objective of this research was to monitor hydraulic throughput and contaminant removal efficacy of operational field 'rain garden' systems. Internationally, rain garden criteria vary widely both from a design and installation perspective (PGC 1993; ARC 2003; GVRD 2005; MDE 2009; SFPUC 2009; Jayaratne et al. 2010). The field rain gardens documented in this research were installed without a topsoil (organic) layer and their effectiveness was unknown as they were not monitored before this research. The reason they were constructed without topsoil was because it was believed that the reduced hydraulic conductivity of topsoil would require a larger areal footprint, which the development could not offer, to treat the same volume of stormwater in the targeted catchment. This research monitored the effectiveness of these systems at mitigating peak flows and reducing contaminant concentrations within the system. The data from this can be used to help inform practitioners about best management strategies for stormwater management.

6.2 Critical Performance Indicators

Effluent pH and related hardness concentrations were identified as critical variables influencing rain garden treatment performance and subsequently potential effluent toxicity.

6.2.1 pH Effect

Strong correlations indicated that pH influences metal speciation, and hence, metal removal capacity in bioinfiltration systems such as rain gardens. The partitioning between the particulate-bound and dissolved metal forms shifts towards the dissolved species below

approximately pH 7 as particulate-bound trace metals are released from particles as free ions (Dempsey et al. 1993; Engstrom 2004). Since particulate fractions are removed more efficiently in biofiltration systems, greater dissolved fractions results in poorer treatment performance (Fletcher et al. 2004). This has profound impacts on bioinfiltration systems designed to treat stormwater as the pH of stormwater in Christchurch was consistently below pH 6.5 S.U. (range of pH 6.0 to 6.8 S.U.) Despite recognizing the effect of pH on metal speciation (Morrison et al. 1990; Dempsey et al. 1993; Landner and Reuther 2004), stormwater systems are not currently designed with pH augmentation. Results from this research demonstrated that effective pH buffering can be accomplished through employing a bulky waste product such as coarsely crushed mussel shells to enhance metal removal in bioinfiltration systems.

6.2.2 Hardness

Hardness has been found to reduce metal ecotoxicity due to increased competition of Ca^{2+} and Mg^{2+} with free metal ions for complexing sites (Hyne et al. 2005). ANZECC guidelines recognize the influence of hardness in hardness modified trigger values (ANZECC 2000), which allow for increased contaminant concentrations at increased water hardness. Mussel shell amendment in the laboratory experiments was found to increase water hardness, thus reducing potential metal ecotoxicity in effluent from the rain gardens should they need to comply with surface water discharge regulations.

6.3 Sustainability

6.3.1 Stormwater Treatment

Stormwater is a major pathway for detaching and transporting urban contaminants into the natural environment and continues to be a leading cause of degradation of urban waterways (Walsh 2000; Egodawatta et al. 2009). Heavy metal concentrations (especially Zn and Cu) in Christchurch stormwater were found to be consistently above the most lenient 80% levels of protection of aquatic species defined in the contextual ANZECC guidelines (Wicke et al. 2009; Wicke et al. 2010).

New Zealand government authorities have focused efforts to educate local communities on the negative impacts of stormwater. Christchurch City Council, the local governing body responsible for stormwater management, have moved from a conveyance off-site approach to

a more on-site infiltrative treatment approach in mitigating stormwater contamination of local waterways (CCC 2003). Rain gardens, a stormwater bioinfiltrative best management practice, can reduce peak flows, minimize outflow volumes and simultaneously reduce contaminant loads to neighbouring waterways. These systems can treat stormwater locally by gravity bioinfiltration and can also recharge groundwater and base-flows in streams through infiltration (Roy et al. 2008). Consistent with the paradigm shift in stormwater management in Christchurch, these natural systems reduce the stress and demand on piped networks and embrace four of the six values outlined in the *Waterways, Wetlands, and Drainage Guide* (CCC 2003), namely, drainage, ecology, culture and landscape. Results from the research reported in this thesis can help provide robust data of rain garden performance based on substrate type and pH effects and hence inform contextual design recommendations.

6.3.2 Locally Available Substrate

Mussel shells are a waste product of the shell fish industry in New Zealand, and are locally available in Christchurch and other parts of New Zealand. Recent research identified mussel shells as a way to mitigate acid mine drainage (McCauley et al. 2009); however, mussel shells have not been used in full scale systems to treat stormwater to date. Results of laboratory experiments indicated mussel shells are a promising rain garden substrate capable of increased metal removal efficiency and decreased toxicity (due to the hardness) of metals discharged from the systems. Using locally available waste products in stormwater treatment systems can reduce the amount of waste going to landfills and offset costs associated with rain garden construction.

6.4 Future Recommendations and Limitations

Additional investigations are recommended to complement, advance and optimize results of this research. Potential areas for additional research include: vegetated laboratory systems, system performance under additional climatic conditions (wet/dry cycles and storm intensity/duration), system lifespan, the use of different waste products to provide pH augmentation and ecotoxicity studies on system effluent.

Two key differences between the mesocosm-scale rain gardens in this research and field-scale rain gardens include vegetation and varying storm intensities. Vegetation was not sustained in laboratory or field systems; however, it is assumed that vegetation would increase removal efficiencies, especially for nutrients, and promote infiltration (Engstrom 2004; Henderson et

al. 2007; Asleson et al. 2009). Varying storm intensity and storm volume may also affect the performance of rain gardens. Large storm events may overwhelm the system and lead to contaminant breakthrough, although a sufficient return period should be incorporated into the design. While this study did not include the effects of vegetation or varying storm events, additional laboratory experiments should be performed to fully investigate the effects of these parameters.

Laboratory systems should be subjected to increased contaminant loading to estimate loading capacity prior to contaminant breakthrough. Additionally, quantifying alkalinity consumption from the mussel shell substrate would help inform the buffering lifespan and could be used to estimate pH potential. While mussel shells are a waste product in New Zealand, and readily available, other waste streams should be investigated for use in stormwater treatment systems where appropriate.

Ecotoxicity studies on system effluent would complement this research and quantify actual impacts on receiving water bodies. The current design standard for rain gardens in New Zealand (TP-10) includes a system under-drain for discharge to surface water; however, this research identified primary discharge to groundwater. The TP-10 guidance was prepared for Auckland conditions (steep topography, predominantly clay soils, 1092 mm annual precipitation (NIWA 2010)), which are different from Christchurch (flat topography, alluvial soils, 653 mm annual precipitation (NIWA 2010)), and thus a more applicable design standard is needed to account for climatic and topographic variation. This updated guidance should integrate different treatment objectives for discharge from treatment systems to surface water or groundwater based on likely impact to the receiving environment.

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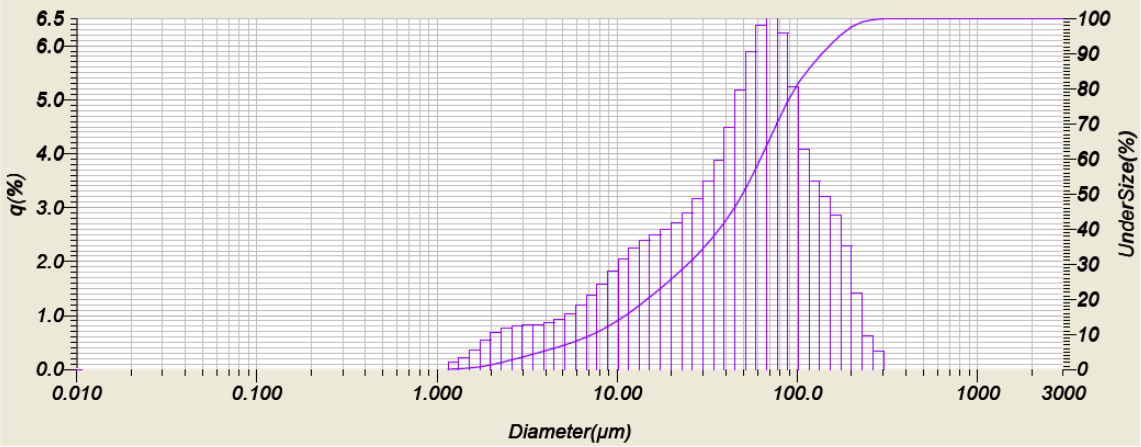
Appendix A- Particle Size Analysis

2011.05.10 14:30:52

HORIBA Laser Scattering Particle Size Distribution Analyzer LA-950

Sample Name	: AHA 100 pH4	Median Size	: 49.60417(µm)
ID#	: 201105101338262	Mean Size	: 61.43597(µm)
Data Name	: Joe topsoil TS1	Std.Dev.	: 52.7841(µm)
Transmittance(R)	: 91.1(%)	Geo.Mean Size	: 38.2267(µm)
Transmittance(B)	: 91.0(%)	Geo.Std.Dev.	: 3.0870(µm)
Circulation Speed	: 7	Mode Size	: 72.1673(µm)
Agitation Speed	: OFF	Span	: OFF
Ultra Sonic	: OFF	Diameter on Cumulative %	: (2)10.00 (%)- 7.3348(µm)
Form of Distribution	: Manual		: (9)90.00 (%)- 137.2042(µm)
Distribution Base	: Volume		
Refractive Index (R)	: 1.70-0.20i(1.33)[1.70-0.20(1.700 - 0.200i),1.33(1.333)]		
Refractive Index (B)	: 1.70-0.20i(1.33)[1.70-0.20(1.700 - 0.200i),1.33(1.333)]		
Material	:		
Source	:		
Lot Number	:		
Test or Assay. Number	: 1		

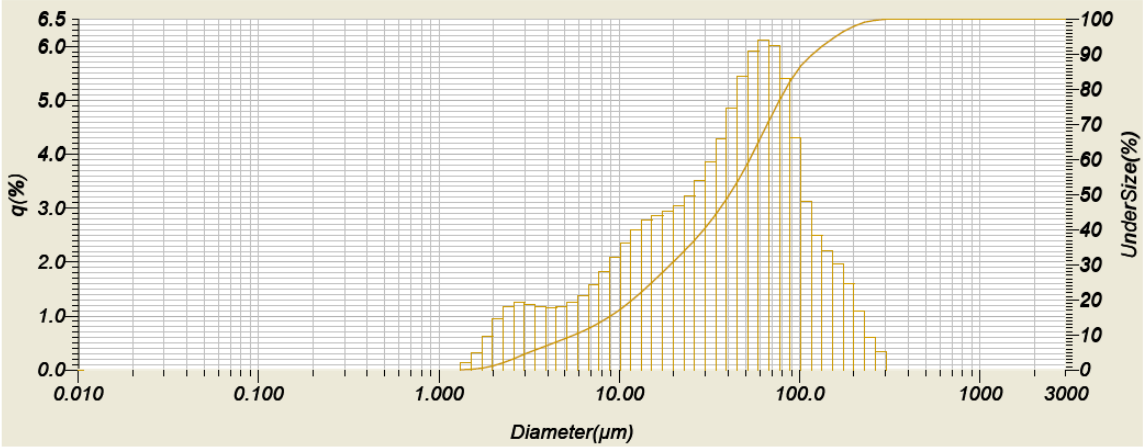


No.	Diameter(µm)	q(%)	UnderSize(%)	No.	Diameter(µm)	q(%)	UnderSize(%)	No.	Diameter(µm)	q(%)	UnderSize(%)	No.	Diameter(µm)	q(%)	UnderSize(%)
1	0.011	0.000	0.000	25	0.296	0.000	0.000	49	7.697	1.368	10.485	73	200.000	2.284	97.632
2	0.013	0.000	0.000	26	0.339	0.000	0.000	50	8.816	1.582	12.067	74	229.075	1.412	99.044
3	0.015	0.000	0.000	27	0.389	0.000	0.000	51	10.097	1.811	13.878	75	262.376	0.615	99.659
4	0.017	0.000	0.000	28	0.445	0.000	0.000	52	11.565	2.037	15.916	76	300.518	0.341	100.000
5	0.020	0.000	0.000	29	0.510	0.000	0.000	53	13.246	2.236	18.152	77	344.206	0.000	100.000
6	0.022	0.000	0.000	30	0.584	0.000	0.000	54	15.172	2.384	20.536	78	394.244	0.000	100.000
7	0.026	0.000	0.000	31	0.669	0.000	0.000	55	17.377	2.488	23.023	79	451.556	0.000	100.000
8	0.029	0.000	0.000	32	0.766	0.000	0.000	56	19.904	2.583	25.606	80	517.200	0.000	100.000
9	0.034	0.000	0.000	33	0.877	0.000	0.000	57	22.797	2.712	28.318	81	592.387	0.000	100.000
10	0.039	0.000	0.000	34	1.005	0.000	0.000	58	26.111	2.905	31.223	82	678.504	0.000	100.000
11	0.044	0.000	0.000	35	1.151	0.000	0.000	59	29.907	3.165	34.388	83	777.141	0.000	100.000
12	0.051	0.000	0.000	36	1.318	0.123	0.123	60	34.255	3.480	37.867	84	890.116	0.000	100.000
13	0.058	0.000	0.000	37	1.510	0.204	0.327	61	39.234	3.883	41.751	85	1019.515	0.000	100.000
14	0.067	0.000	0.000	38	1.729	0.351	0.678	62	44.938	4.477	46.227	86	1167.725	0.000	100.000
15	0.076	0.000	0.000	39	1.981	0.530	1.209	63	51.471	5.184	51.411	87	1337.481	0.000	100.000
16	0.087	0.000	0.000	40	2.269	0.679	1.888	64	58.953	5.884	57.294	88	1531.914	0.000	100.000
17	0.100	0.000	0.000	41	2.599	0.769	2.657	65	67.523	6.367	63.662	89	1754.613	0.000	100.000
18	0.115	0.000	0.000	42	2.976	0.806	3.463	66	77.339	6.590	70.252	90	2009.687	0.000	100.000
19	0.131	0.000	0.000	43	3.409	0.817	4.280	67	88.583	6.236	76.487	91	2301.841	0.000	100.000
20	0.150	0.000	0.000	44	3.905	0.829	5.109	68	101.460	5.234	81.721	92	2636.467	0.000	100.000
21	0.172	0.000	0.000	45	4.472	0.862	5.971	69	116.210	4.077	85.798	93	3000.000	0.000	100.000
22	0.197	0.000	0.000	46	5.122	0.929	6.900	70	133.103	3.487	89.285				
23	0.226	0.000	0.000	47	5.867	1.035	7.935	71	152.453	3.197	92.483				
24	0.259	0.000	0.000	48	6.720	1.182	9.118	72	174.616	2.865	95.348				

2011.05.10 14:37:39

HORIBA Laser Scattering Particle Size Distribution Analyzer LA-950

Sample Name	: AHA 100 pH4	Median Size	: 40.79592(µm)
ID#	: 201105101421265	Mean Size	: 53.26085(µm)
Data Name	: 201105101421265	Std.Dev.	: 49.5508(µm)
Transmittance(R)	: 91.4(%)	Geo.Mean Size	: 31.9791(µm)
Transmittance(B)	: 90.9(%)	Geo.Std.Dev.	: 3.1477(µm)
Circulation Speed	: 7	Mode Size	: 63.1538(µm)
Agitation Speed	: OFF	Span	: OFF
Ultra Sonic	: OFF	Diameter on Cumulative %	: (2)10.00 (%)- 5.6659(µm)
Form of Distribution	: Manual		: (9)90.00 (%)- 117.8436(µm)
Distribution Base	: Volume		
Refractive Index (R)	: 1.40-0.50i(1.33)[1.40-0.50(1.400 - 0.500i),1.33(1.333)]		
Refractive Index (B)	: 1.40-0.50i(1.33)[1.40-0.50(1.400 - 0.500i),1.33(1.333)]		
Material	:		
Source	:		
Lot Number	:		
Test or Assay. Number	: 1		



No.	Diameter(µm)	q(%)	UnderSize(%)	No.	Diameter(µm)	q(%)	UnderSize(%)	No.	Diameter(µm)	q(%)	UnderSize(%)	No.	Diameter(µm)	q(%)	UnderSize(%)
1	0.011	0.000	0.000	25	0.296	0.000	0.000	49	7.697	1.575	13.272	73	200.000	1.600	97.988
2	0.013	0.000	0.000	26	0.339	0.000	0.000	50	8.816	1.816	15.088	74	229.075	1.089	99.077
3	0.015	0.000	0.000	27	0.389	0.000	0.000	51	10.097	2.083	17.171	75	262.376	0.594	99.670
4	0.017	0.000	0.000	28	0.445	0.000	0.000	52	11.565	2.355	19.527	76	300.518	0.330	100.000
5	0.020	0.000	0.000	29	0.510	0.000	0.000	53	13.246	2.599	22.125	77	344.206	0.000	100.000
6	0.022	0.000	0.000	30	0.584	0.000	0.000	54	15.172	2.769	24.894	78	394.244	0.000	100.000
7	0.026	0.000	0.000	31	0.669	0.000	0.000	55	17.377	2.866	27.760	79	451.556	0.000	100.000
8	0.029	0.000	0.000	32	0.766	0.000	0.000	56	19.904	2.937	30.697	80	517.200	0.000	100.000
9	0.034	0.000	0.000	33	0.877	0.000	0.000	57	22.797	3.043	33.740	81	592.387	0.000	100.000
10	0.039	0.000	0.000	34	1.005	0.000	0.000	58	26.111	3.228	36.968	82	678.504	0.000	100.000
11	0.044	0.000	0.000	35	1.151	0.000	0.000	59	29.907	3.503	40.471	83	777.141	0.000	100.000
12	0.051	0.000	0.000	36	1.318	0.000	0.000	60	34.255	3.852	44.323	84	890.116	0.000	100.000
13	0.058	0.000	0.000	37	1.510	0.124	0.124	61	39.234	4.283	48.606	85	1019.515	0.000	100.000
14	0.067	0.000	0.000	38	1.729	0.308	0.432	62	44.938	4.848	53.454	86	1167.725	0.000	100.000
15	0.076	0.000	0.000	39	1.981	0.618	1.050	63	51.471	5.438	58.891	87	1337.481	0.000	100.000
16	0.087	0.000	0.000	40	2.269	0.944	1.994	64	58.953	5.916	64.808	88	1531.914	0.000	100.000
17	0.100	0.000	0.000	41	2.599	1.164	3.158	65	67.523	6.101	70.909	89	1754.613	0.000	100.000
18	0.115	0.000	0.000	42	2.976	1.239	4.397	66	77.339	6.003	76.912	90	2009.687	0.000	100.000
19	0.131	0.000	0.000	43	3.409	1.216	5.613	67	88.583	5.405	82.317	91	2301.841	0.000	100.000
20	0.150	0.000	0.000	44	3.905	1.167	6.780	68	101.460	4.304	86.620	92	2636.467	0.000	100.000
21	0.172	0.000	0.000	45	4.472	1.139	7.919	69	116.210	3.124	89.744	93	3000.000	0.000	100.000
22	0.197	0.000	0.000	46	5.122	1.160	9.079	70	133.103	2.488	92.232				
23	0.226	0.000	0.000	47	5.867	1.239	10.318	71	152.453	2.198	94.431				
24	0.259	0.000	0.000	48	6.720	1.379	11.697	72	174.616	1.957	96.388				

Appendix B- Substrate Baseline Laboratory Analytical Reports

Hill Laboratories
BETTER TESTING BETTER RESULTS

R J Hill Laboratories Limited
1 Clyde Street
Private Bag 3205
Hamilton 3240, New Zealand

Tel +64 7 858 2000
Fax +64 7 858 2001
Email mail@hill-labs.co.nz
Web www.hill-labs.co.nz

ANALYSIS REPORT

Client:	Department of Civil Engineering	**Lab No:**	829470	SPv1
Contact:	Dr A O'Sullivan	**Date Registered:**	23-Sep-2010	
	C/- Department of Civil Engineering	**Date Reported:**	06-Oct-2010	
	University of Canterbury	**Quote No:**	41288	
	Private Bag 4800	**Order No:**	501292	
	CHRISTCHURCH 8140	**Client Reference:**	Joe Good Stormwater	
		Submitted By:	JF Good	

Sample Type: Miscellaneous

Sample Name:		BM1-22-09-2010 22-Sep-2010 9:30 am	BM2-22-09-2010 22-Sep-2010 9:35 am	TS1-22-09-2010 22-Sep-2010 9:45 am	TS2-22-09-2010 22-Sep-2010 9:50 am	
Lab Number:		829470.1	829470.2	829470.3	829470.4	
TCLP Weight of Sample Taken	g	60	60	100	100	-
TCLP Initial Sample pH	pH Units	5.1	5.4	5.8	5.8	-
TCLP Acid Adjusted Sample pH	pH Units	1.4	1.4	1.5	1.5	-
TCLP Extraction Fluid Description*		NaOH/Acetic acid at pH 4.93 +/- 0.05	NaOH/Acetic acid at pH 4.93 +/- 0.05	NaOH/Acetic acid at pH 4.93 +/- 0.05	NaOH/Acetic acid at pH 4.93 +/- 0.05	-
TCLP Extraction Fluid pH	pH Units	4.9	4.9	4.9	4.9	-
TCLP Post Extraction Sample pH	pH Units	4.9	4.9	4.9	4.9	-
Total Recoverable Copper	mg/kg dry wt	< 4	< 4	7	7	-
Total Recoverable Lead	mg/kg dry wt	6.2	6.4	15.2	15.8	-
Total Recoverable Zinc	mg/kg dry wt	19	22	49	49	-
Total Nitrogen	g/100g dry wt	0.34	0.33	0.20	0.23	-

Sample Type: Aqueous

Sample Name:		BM1-22-09-2010 (TCLP Extract)	BM2-22-09-2010 (TCLP Extract)	TS1-22-09-2010 (TCLP Extract)	TS2-22-09-2010 (TCLP Extract)	
Lab Number:		829470.5	829470.6	829470.7	829470.8	
Total Copper	g/m³	< 0.011	< 0.011	< 0.011	< 0.011	-
Total Lead	g/m³	0.003	0.003	0.004	0.003	-
Total Zinc	g/m³	0.064	0.065	< 0.021	< 0.021	-

SUMMARY OF METHODS

The following table(s) gives a brief description of the methods used to conduct the analyses for this job. The detection limits given below are those attainable in a relatively clean matrix. Detection limits may be higher for individual samples should insufficient sample be available, or if the matrix requires that dilutions be performed during analysis.

Sample Type: Miscellaneous

Test	Method Description	Default Detection Limit	Samples
Individual Tests			
Environmental Solids Sample Preparation	Air dried at 35°C and sieved, <2mm fraction.	-	1-4
Total Recoverable digestion	Nitric / hydrochloric acid digestion. US EPA 200.2.	-	1-4
Total Recoverable Copper	Dried sample, sieved as specified (if required). Nitric/Hydrochloric acid digestion, ICP-MS, screen level. US EPA 200.2.	2 mg/kg dry wt	1-4
Total Recoverable Lead	Dried sample, sieved as specified (if required). Nitric/Hydrochloric acid digestion, ICP-MS, screen level. US EPA 200.2.	0.4 mg/kg dry wt	1-4
Total Recoverable Zinc	Dried sample, sieved as specified (if required). Nitric/Hydrochloric acid digestion, ICP-MS, screen level. US EPA 200.2.	4 mg/kg dry wt	1-4
Total Nitrogen	Catalytic Combustion (900°C, O_2), separation, Thermal Conductivity Detector [Elementar Analyser].	0.05 g/100g dry wt	1-4
TCLP Profile*	Extraction at 30 +/- 2 rpm for 18 +/- 2 hours, (Ratio 1g sample : 20g extraction fluid). US EPA 1311	-	1-4

This Laboratory is accredited by International Accreditation New Zealand (IANZ), which represents New Zealand in the International Laboratory Accreditation Cooperation (ILAC). Through the ILAC Mutual Recognition Arrangement (ILAC-MRA) this accreditation is internationally recognised.
The tests reported herein have been performed in accordance with the terms of accreditation, with the exception of tests marked *, which are not accredited.

Sample Type: Miscellaneous			
Test	**Method Description**	**Default Detection Limit**	**Samples**
TCLP Profile			
TCLP Weight of Sample Taken	Gravimetric. US EPA 1311.	0.1 g	1-4
TCLP Initial Sample pH	pH meter. US EPA 1311.	0.1 pH Units	1-4
TCLP Acid Adjusted Sample pH	pH meter. US EPA 1311.	0.1 pH Units	1-4
TCLP Extractant Type*	US EPA 1311.	-	1-4
TCLP Extraction Fluid pH	pH meter. US EPA 1311.	0.1 pH Units	1-4
TCLP Post Extraction Sample pH	pH meter. US EPA 1311.	0.1 pH Units	1-4

Sample Type: Aqueous			
Test	**Method Description**	**Default Detection Limit**	**Samples**
Individual Tests			
Total Digestion of Extracted Samples*	Nitric acid digestion. APHA 3030 E 21st ed. 2005.	-	5-8
Total Copper	Nitric acid digestion, ICP-MS, screen level. APHA 3125 B 21st ed. 2005.	0.011 g/m^3	5-8
Total Lead	Nitric acid digestion, ICP-MS, screen level. APHA 3125 B 21st ed. 2005.	0.0021 g/m^3	5-8
Total Zinc	Nitric acid digestion, ICP-MS, screen level. APHA 3125 B 21st ed. 2005.	0.021 g/m^3	5-8

These samples were collected by yourselves (or your agent) and analysed as received at the laboratory.

Samples are held at the laboratory after reporting for a length of time depending on the preservation used and the stability of the analytes being tested. Once the storage period is completed the samples are discarded unless otherwise advised by the client.

Ara Heron BSc (Tech)
Client Services Manager - Environmental Division

Hill Laboratories
BETTER TESTING BETTER RESULTS

R J Hill Laboratories Limited
1 Clyde Street
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Hamilton 3240, New Zealand

Tel +64 7 858 2000
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Web www.hill-labs.co.nz

ANALYSIS REPORT

Client:	Department of Civil Engineering	**Lab No:**	937489	SPv2
Contact:	JF Good	**Date Registered:**	28-Sep-2011	
	C/- Department of Civil Engineering	**Date Reported:**	13-Oct-2011	
	University of Canterbury	**Quote No:**	46475	
	Private Bag 4800	**Order No:**	512576	
	CHRISTCHURCH 8140	**Client Reference:**	Purchase Order 512576	
		Submitted By:	JF Good	

Sample Type: Sediment

Sample Name:		GS1 27-Sep-2011	GS2 27-Sep-2011			
Lab Number:		937489.3	937489.4			
TCLP Weight of Sample Taken	g	100	100	-	-	-
TCLP Initial Sample pH	pH Units	6.4	6.0	-	-	-
TCLP Acid Adjusted Sample pH	pH Units	1.4	1.4	-	-	-
TCLP Extractant Type*		NaOH/Acetic acid at pH 4.93 +/- 0.05	NaOH/Acetic acid at pH 4.93 +/- 0.05	-	-	-
TCLP Extraction Fluid pH	pH Units	4.9	4.9	-	-	-
TCLP Post Extraction Sample pH	pH Units	4.9	4.9	-	-	-
Total Recoverable Copper	mg/kg dry wt	5	5	-	-	-
Total Recoverable Lead	mg/kg dry wt	6.9	6.6	-	-	-
Total Recoverable Zinc	mg/kg dry wt	31	31	-	-	-
Total Nitrogen*	g/100g dry wt	< 0.05	< 0.05	-	-	-

Sample Type: Miscellaneous

Sample Name:		MS1 27-Sep-2011	MS2 27-Sep-2011			
Lab Number:		937489.1	937489.2			
TCLP Weight of Sample Taken*	g	100	100	-	-	-
TCLP Initial Sample pH*	pH Units	8.5	8.4	-	-	-
TCLP Acid Adjusted Sample pH*	pH Units	5.3	2.5	-	-	-
TCLP Extractant Type*		Acetic acid solution at pH 2.88 +/- 0.05	NaOH/Acetic acid at pH 4.93 +/- 0.05	-	-	-
TCLP Extraction Fluid pH*	pH Units	2.9	4.9	-	-	-
TCLP Post Extraction Sample pH*	pH Units	6.2	6.5	-	-	-
Total Recoverable Copper*	mg/kg dry wt	< 4	< 4	-	-	-
Total Recoverable Lead*	mg/kg dry wt	< 0.8	< 0.8	-	-	-
Total Recoverable Zinc*	mg/kg dry wt	< 8	< 8	-	-	-
Total Nitrogen*	g/100g dry wt	1.69	1.72	-	-	-

Sample Type: Aqueous

Sample Name:		MS1 [TCLP Extract]	MS2 [TCLP Extract]	GS1 [TCLP Extract]	GS2 [TCLP Extract]	
Lab Number:		937489.5	937489.6	937489.7	937489.8	
Total Copper	g/m³	0.013	< 0.011	< 0.011	< 0.011	-
Total Lead	g/m³	< 0.0021	< 0.0021	0.0024	0.0164	-
Total Zinc	g/m³	0.036	< 0.021	< 0.021	0.057	-

Analyst's Comments

It should be noted that the mussel shells samples (937489.1-2) were dried and crushed before the before TCLP extraction procedure was carried out.

SUMMARY OF METHODS

The following table(s) gives a brief description of the methods used to conduct the analyses for this job. The detection limits given below are those attainable in a relatively clean matrix. Detection limits may be higher for individual samples should insufficient sample be available, or if the matrix requires that dilutions be performed during analysis.

Sample Type: Miscellaneous			
Test	**Method Description**	**Default Detection Limit**	**Samples**
Individual Tests			
Environmental Solids Sample Preparation*	Air dried at 35°C and sieved, <2mm fraction. Used for sample preparation. May contain a residual moisture content of 2-5%.	-	1-4
Rockgrind*	Crush to ~2mm using Boyd Rock Crusher MarkIII.	-	1-2
Total Recoverable digestion*	Nitric / hydrochloric acid digestion. US EPA 200.2.	-	1-4
Total Recoverable Copper*	Dried sample, sieved as specified (if required). Nitric/Hydrochloric acid digestion, ICP-MS, screen level. US EPA 200.2.	2 mg/kg dry wt	1-4
Total Recoverable Lead*	Dried sample, sieved as specified (if required). Nitric/Hydrochloric acid digestion, ICP-MS, screen level. US EPA 200.2.	0.4 mg/kg dry wt	1-4
Total Recoverable Zinc*	Dried sample, sieved as specified (if required). Nitric/Hydrochloric acid digestion, ICP-MS, screen level. US EPA 200.2.	4 mg/kg dry wt	1-4
Total Nitrogen*	Catalytic Combustion (900°C, O_2), separation, Thermal Conductivity Detector [Elementar Analyser].	0.05 g/100g dry wt	1-4
TCLP Profile*	Extraction at 30 +/- 2 rpm for 18 +/- 2 hours, (Ratio 1g sample : 20g extraction fluid). US EPA 1311	-	1-4
TCLP Profile			
TCLP Weight of Sample Taken*	Gravimetric. US EPA 1311.	0.1 g	1-4
TCLP Initial Sample pH*	pH meter. US EPA 1311.	0.1 pH Units	1-4
TCLP Acid Adjusted Sample pH*	pH meter. US EPA 1311.	0.1 pH Units	1-4
TCLP Extractant Type*	US EPA 1311.	-	1-4
TCLP Extraction Fluid pH*	pH meter. US EPA 1311.	0.1 pH Units	1-4
TCLP Post Extraction Sample pH*	pH meter. US EPA 1311.	0.1 pH Units	1-4

Sample Type: Aqueous			
Test	**Method Description**	**Default Detection Limit**	**Samples**
Individual Tests			
Total Digestion of Extracted Samples*	Nitric acid digestion. APHA 3030 E 21st ed. 2005.	-	5-8
Total Copper	Nitric acid digestion, ICP-MS, screen level. APHA 3125 B 21st ed. 2005.	0.011 g/m³	5-8
Total Lead	Nitric acid digestion, ICP-MS, screen level. APHA 3125 B 21st ed. 2005.	0.0021 g/m³	5-8
Total Zinc	Nitric acid digestion, ICP-MS, screen level. APHA 3125 B 21st ed. 2005.	0.021 g/m³	5-8

These samples were collected by yourselves (or your agent) and analysed as received at the laboratory.

Samples are held at the laboratory after reporting for a length of time depending on the preservation used and the stability of the analytes being tested. Once the storage period is completed the samples are discarded unless otherwise advised by the client.

Ara Heron BSc (Tech)
Client Services Manager - Environmental Division

Appendix C- Addington Rain Garden As-Built Drawings

RAIN GARDEN

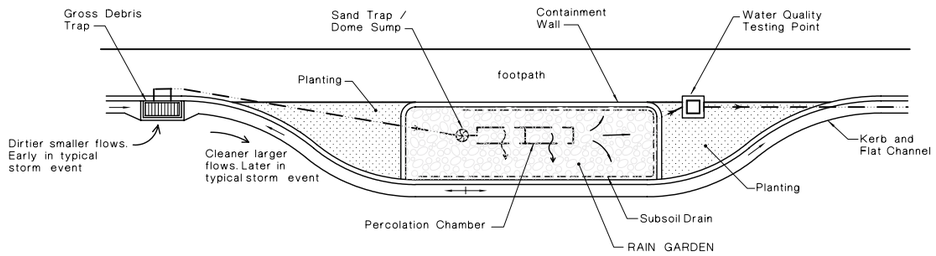
TYPICAL COMPONENTS AND FEATURES

The stormwater which flows in road side gutters comes from both the roadway and private property. This water discharges eventually into waterways and rivers and is a source of pollution. Options are being explored to improve the quality of this stormwater.

Best stormwater management practises adopt a 'treatment train approach' where various size groups of suspended particles are progressively targeted.

Flows early in a typical rain storm event are dirtier than larger later flows. It is this first early flow that the 'Rain Gardens' capture/treat. Larger flows will bypass the Rain Garden.

The Rain Gardens being trialled in Addington rely on several filtration stages and sedimentation stages to improve stormwater quality. Regular maintenance is needed for their continued functioning. Several variations on this basic theme are being trialled.



PLAN
1:100

GROSS DEBRIS TRAP

- Acts as a two stage filter and sedimentation chamber
- Large (Gross) debris trapped on surface
- Meg450 unit and twin drilled end caps trap 10-20mm particles
- Long duration ponding in road channel indicates maintenance of GDT may be necessary. GDT may need emptying and Meg450 sump filter unit cleaned.

SUBMERGED INLET PIPE

- Conveys stormwater to Rain Garden
- When sand trap / dome sump full above inlet pipe, pipe will block causing minor ponding in roadway.
- Long duration ponding in road channel indicates maintenance of Dome Sump / Sand Trap may be necessary. A suction truck may need to remove sand etc from Dome Sump.

SAND TRAP / DOME SUMP

- Acts as a sedimentation tank allowing sand and grit to settle / collect.
- An inspection point for filter bed functioning.
- An access point for suction truck hose.
- Outlet for larger flows to allow horizontal dispersal of stormwater.

CONTAINMENT WALLS

- Protects shallow road subbase from weakening by preventing saturation of subbase.
- Support and protection for Meg450 subsoil panel drains
- Support of adjacent land and services during periodic filter medium replacement.

PLANTING

- Aesthetics; designed by Landscape Architect.
- Bioremediation; processes some pollutants in stormwater.

PERCOLATION CHAMBERS

- Disperses flows over a wide surface area.
- Collect / trap sludge by both filtration through filter cloth and by sedimentation.
- Provides detention volume of stormwater prior to soil filtration.
- Long duration ponding in road channel indicates maintenance of dispersal chamber may be necessary. Mulch needs to be raked back to expose removable lids of 1.5m versitanks. Water may be needed to dislodge sludge to allow suction hose to remove sludge.

GRAVEL MULCH (Typically washed gravel)

- Complement Planting / Landscape feature.
- Void space used to store detained stormwater.
- Camouflage percolation chambers/lids.

FILTER FABRIC

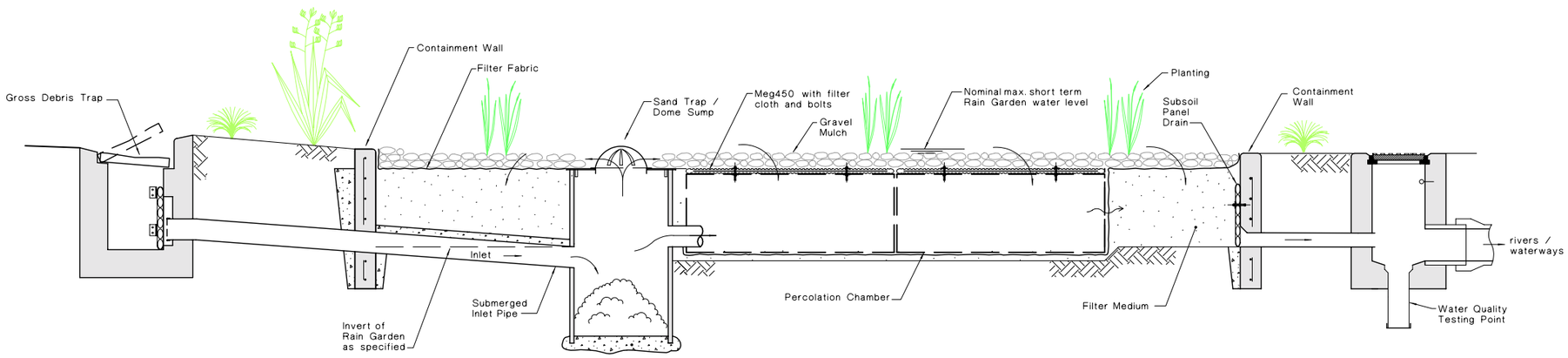
- Removes sludge before it enters filter medium.

FILTER MEDIUM (Typically sand)

- Filters fine suspended particles
- Storage volume for detained stormwater
- Planting medium
- Long duration ponding in road channel may indicate maintenance of filter medium necessary. Initial efforts should only require filter medium to be aerated with sideways movement and frequent penetration by a hand fork or similar. When this becomes ineffective a check should be then made to check functioning of Meg450 filter. If Meg450 still functions, filter medium and planting will need replacement.

SUBSOIL PANEL DRAIN

- Transports and collects filtered stormwater before stormwater discharged to pipe lines, which ultimately empty into our rivers and waterways. Typically Meg450.



SECTION
1:20

ISSUE	AMENDMENTS	SIGNED	DATE

CITYSOLUTIONS

DATUM	C.D.B.
BENCH MK.	
SURVEY FB	
SURVEY LB	
CONSTN.EB	
CONSTN.LB	
DESIGN FILE	

	NAME	SIGNED	DATE
DESIGNED	P. Wehrmann		11/05
DES. REVIEW			
DRAWN	D. Mitchell		11/05
DRW. CHECK			

APPROVED

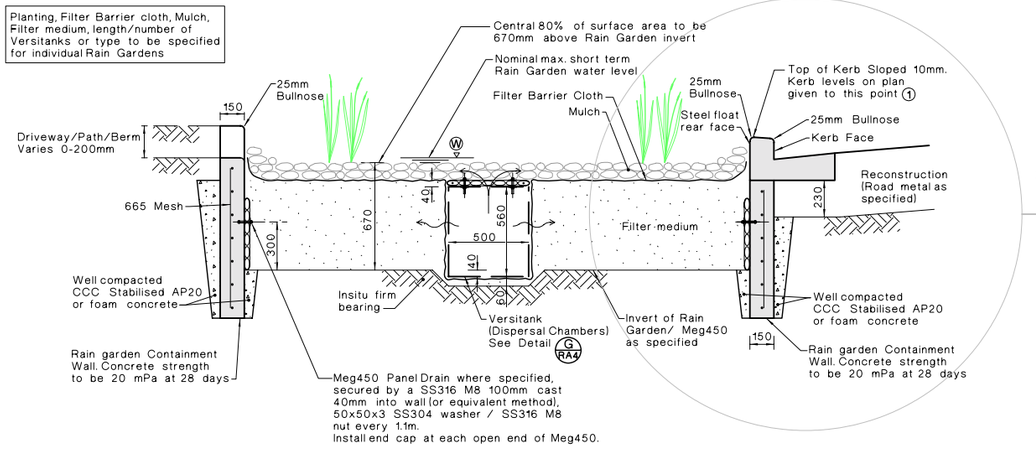
DATE

CLIENT: CHRISTCHURCH CITY COUNCIL · YOUR PEOPLE · YOUR CITY

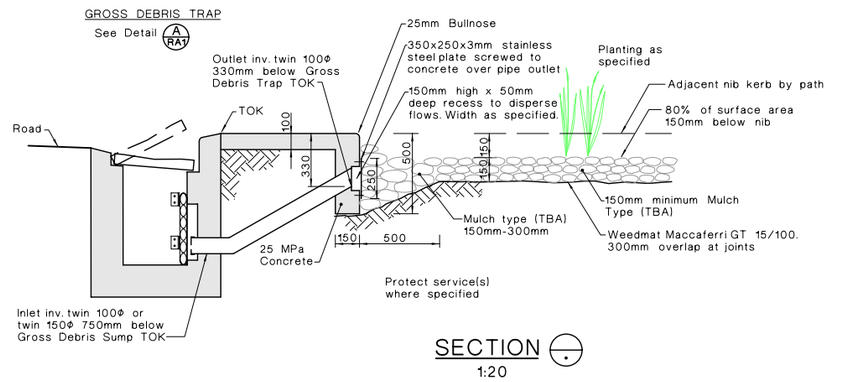
PROJECT TITLE: RAIN GARDEN

DRAWING TITLE: TYPICAL COMPONENTS AND FEATURES

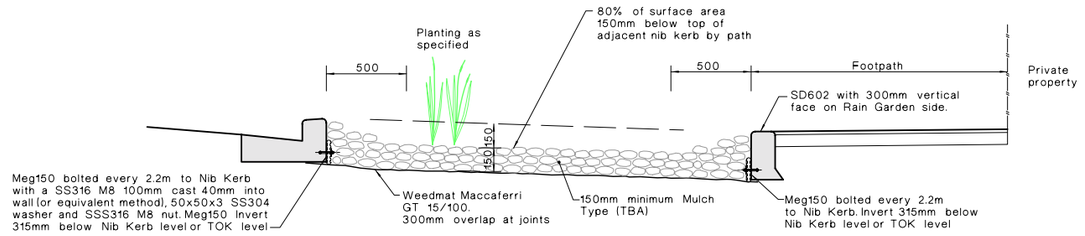
CONTRACT NUMBER	ORIGINAL SHEET SIZE	SCALES
05/06-	A1	1:20 1:100
FILE REFERENCE		
542/920		
DRAWING NUMBER	SHEET	
RD1598	RA5 OF	



TYPICAL CROSS SECTION
1:20



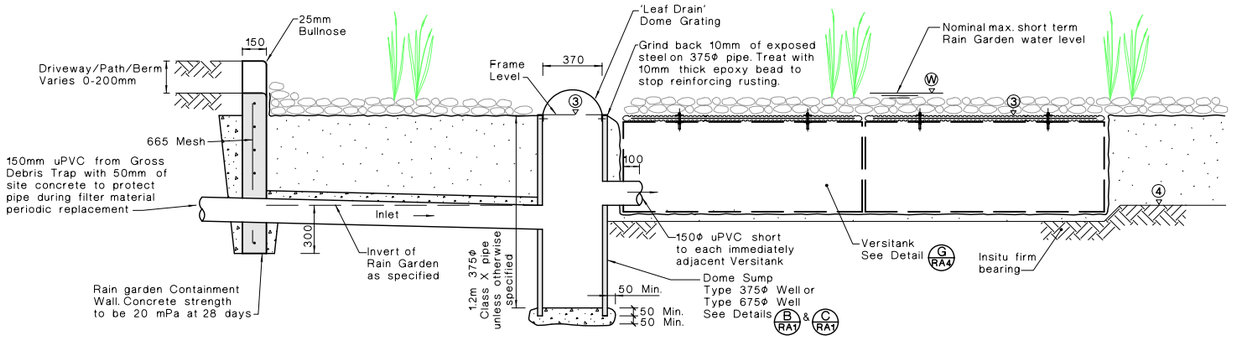
SECTION
1:20



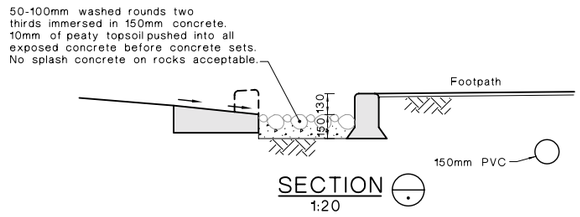
SECTION
1:20

DOME SUMP SAND SUCTION POINT/SAND TRAP

Dome Sump Lid to be no further than 2.0m from kerb face to allow suction truck access.



INLET AND SILT TRAP DETAIL
1:20

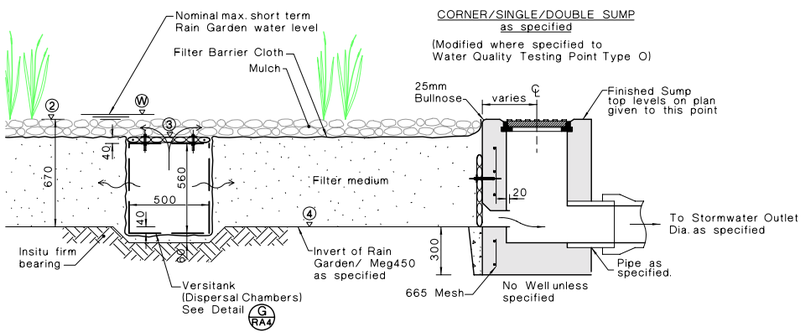


SECTION
1:20

RAIN GARDEN - TYPE 2

NOTE

SEE SPECIFIC DESIGN LEVEL LIST ASSOCIATED WITH EACH RAIN GARDEN. LEVELS SHOWN - (1)

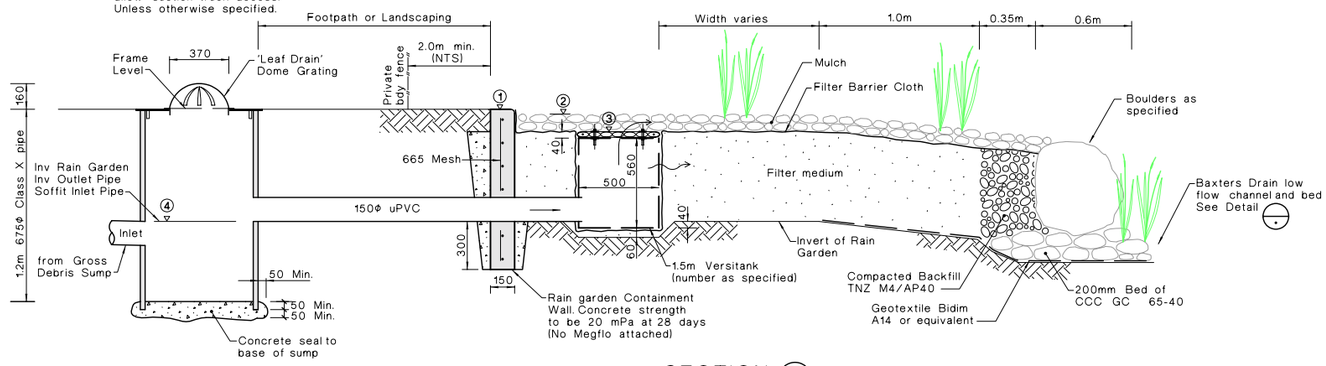


OUTLET DETAIL / INSPECTION POINT
1:20

RAIN GARDEN - TYPE 1

DOME SUMP SAND SUCTION POINT/SAND TRAP

Dome Sump Lid to be no further than 2.0m from kerb face to allow suction truck access. Unless otherwise specified.



SECTION
1:20

RAIN GARDEN - TYPE 3

ISSUE	AMENDMENTS	SIGNED	DATE

CITYSOLUTIONS

DATUM	C.D.B.
BENCH MK.	
SURVEY FB.	
SURVEY LB.	
CONSTN.EB.	
CONSTN.LB.	
DESIGN FILE	

	NAME	SIGNED	DATE
DESIGNED	P. Wehrmann		10/05
DES. REVIEW			
DRAWN	D. Mitchell		10/05
DRW. CHECK			

APPROVED

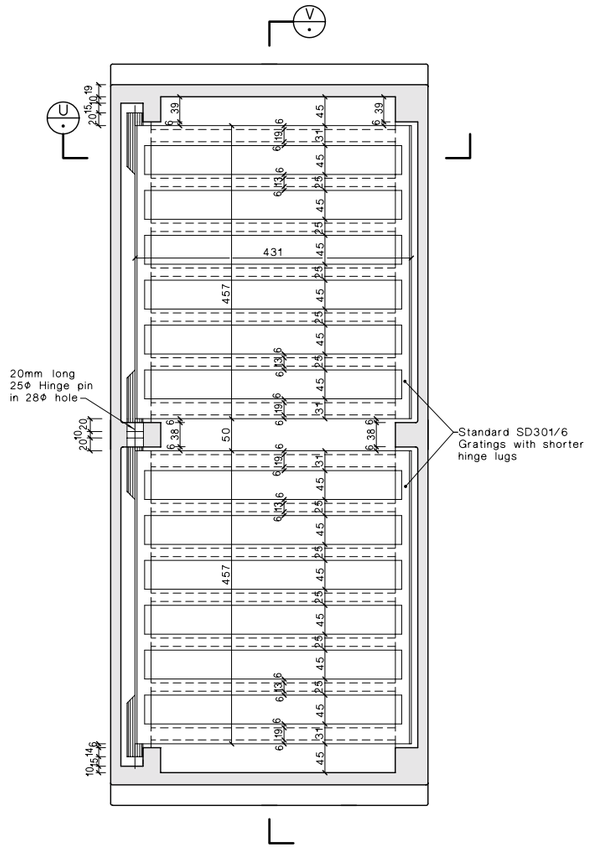
DATE

CLIENT: CHRISTCHURCH CITY COUNCIL · YOUR PEOPLE · YOUR CITY

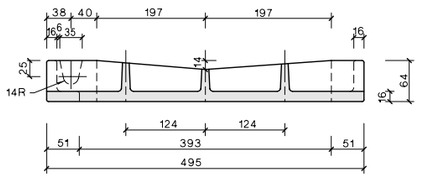
PROJECT TITLE: RAIN GARDENS

DRAWING TITLE: SECTIONS

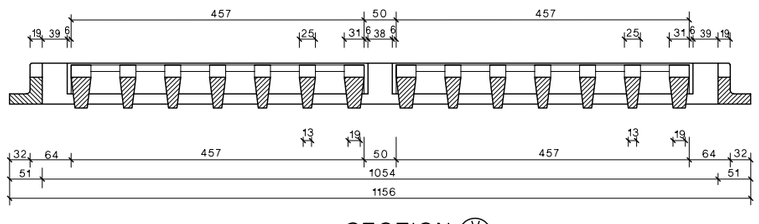
CONTRACT NUMBER	ORIGINAL SHEET SIZE	SCALES
05/06-	A1	1:20
FILE REFERENCE: 542/920		
DRAWING NUMBER: RD1598	SHEET: RA2 OF	



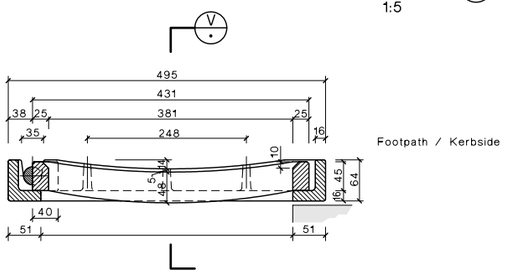
GROSS DEBRIS TRAP GRATING PLAN (W)
1:5



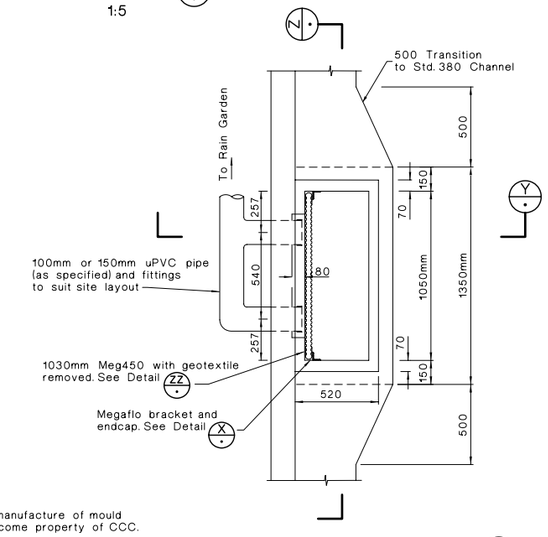
FRAME ELEVATION
1:5



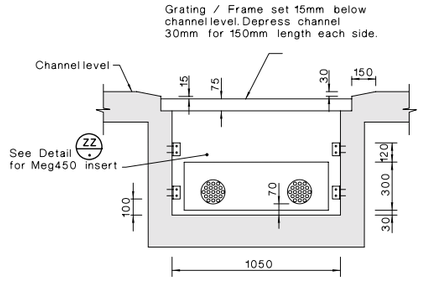
SECTION (V)
1:5



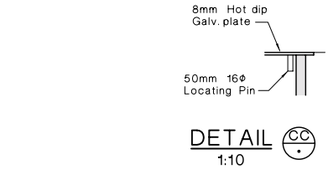
SECTION (U)
1:5



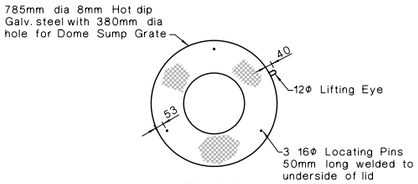
GROSS DEBRIS TRAP PLAN (A)
1:20
(GRATE AND MEGFLO FILTER NOT SHOWN FOR CLARITY)



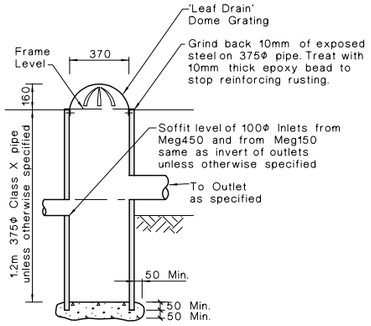
SECTION (Z)
1:20



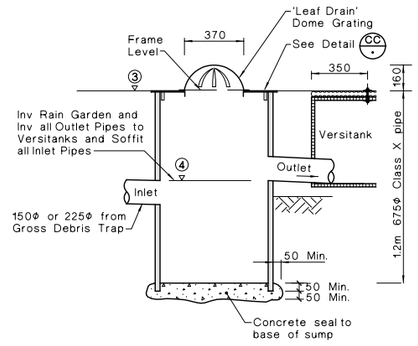
DETAIL (CC)
1:10



PLAN



DOME SUMP TYPE 375 WELL (B)
1:20

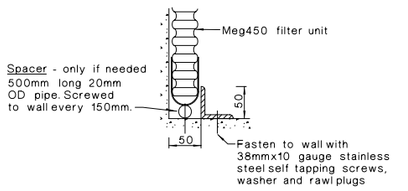


DOME SUMP TYPE 675 WELL (C)
1:20

FRAME ELEVATION
1:5

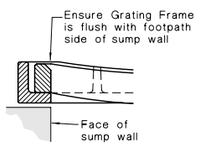
NOTES

1. Contractor to arrange manufacture of mould for casting. Mould to become property of CCC.
2. Yield points shall be provided in accordance with plan SD341 except that in all cases two yield points and on short pipe shall be used.
3. General method of pipe laying, angle connections, corbels, etc. to be in accordance with plan SD341.
4. Cast Iron frame and grate to be in accordance with CSS.

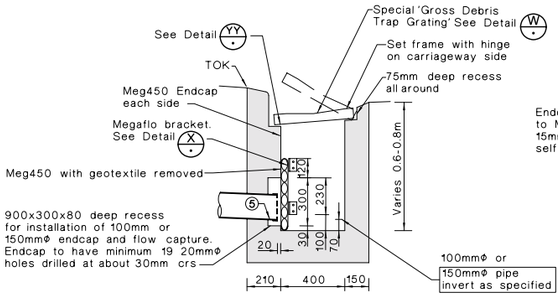


50x50x6 Stainless steel angle iron cut into 80mm lengths. One 7mm hole 20mm from each end to fasten to trap wall. Ensure final fitting of stainless steel brackets and modified twin SD601/6 frames allows easy replacement / cleaning / removal of 450 Megaflo and attached endcaps.

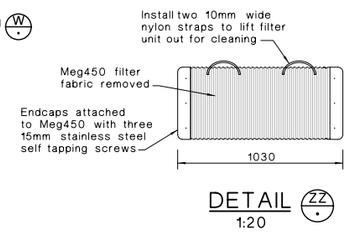
MEGAFLO BRACKET (X)
1:5



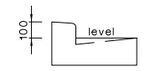
DETAIL (YY)
1:5



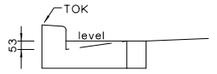
SECTION (Y)
1:20



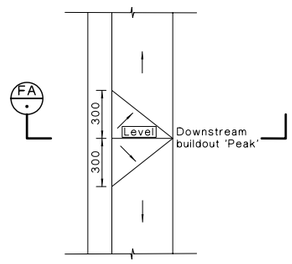
DETAIL (ZZ)
1:20



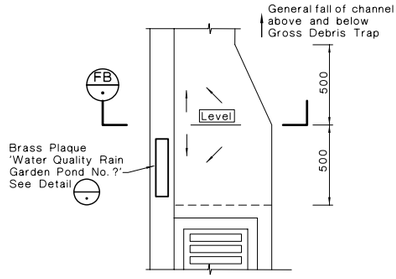
SECTION (FA)
1:20



SECTION (FB)
1:20



1. Where Gross Debris Trap is in a shallow valley



2. Where Gross Debris Trap positioned on constant grade Kerb and Flat Channel

KERB BUND (F)
1:20

ISSUE	AMENDMENTS	SIGNED	DATE

CITYSOLUTIONS

DATUM	C.D.B.
BENCH MK.	
SURVEY FB	
SURVEY LB	
CONSTN. EB	
CONSTN. LB	
DESIGN FILE	

	NAME	SIGNED	DATE
DESIGNED	P. Wehrmann		10/05
DES. REVIEW			
DRAWN	D. Mitchell		10/05
DRW. CHECK			

APPROVED

DATE

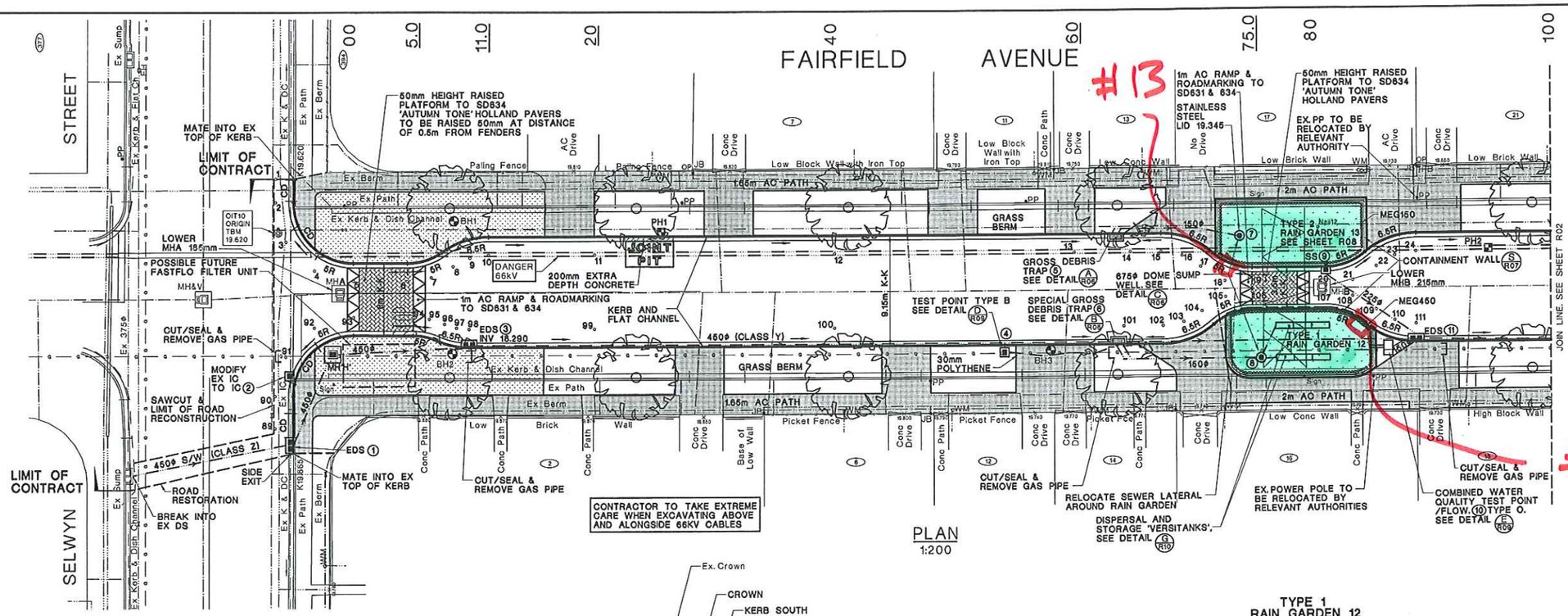
CLIENT: CHRISTCHURCH CITY COUNCIL · YOUR PEOPLE · YOUR CITY

PROJECT TITLE: ADDINGTON CLUSTER RAIN GARDEN

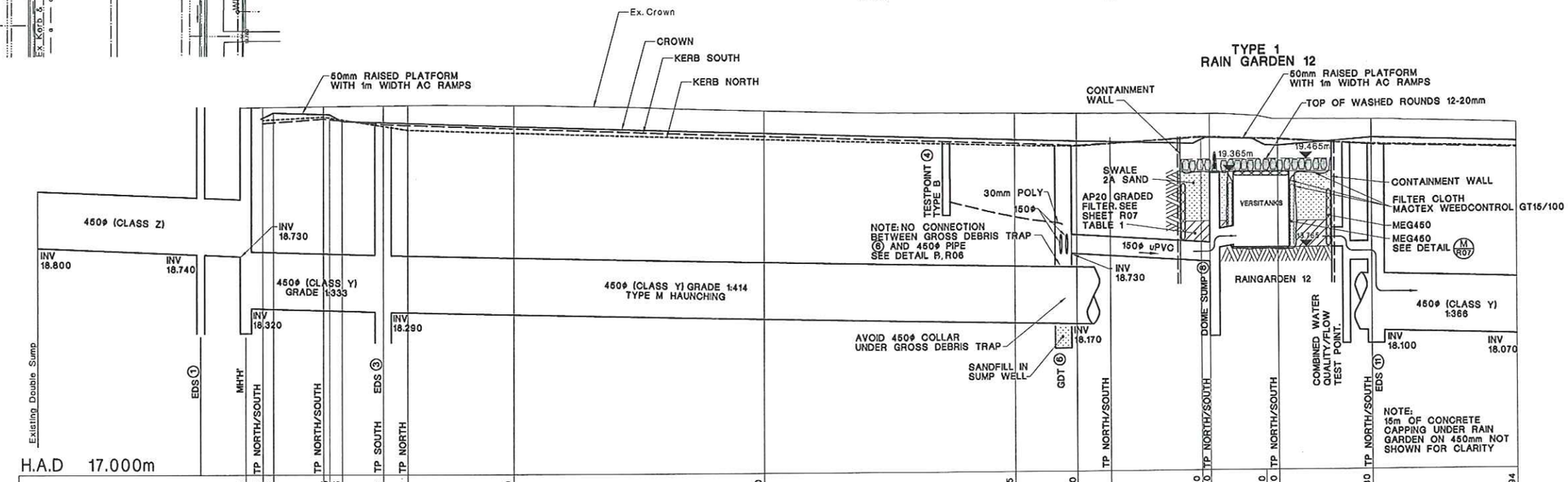
DRAWING TITLE: DETAILS

CONTRACT NUMBER	ORIGINAL SHEET SIZE	SCALES
05/06-	A1	As Shown
FILE REFERENCE		
DRAWING NUMBER	SHEET	
RD1598	RA1 OF	

FAIRFIELD AVENUE



PLAN
1:200



H.A.D 17.000m

| | | |

Appendix D- Weir Stage-Discharge Data

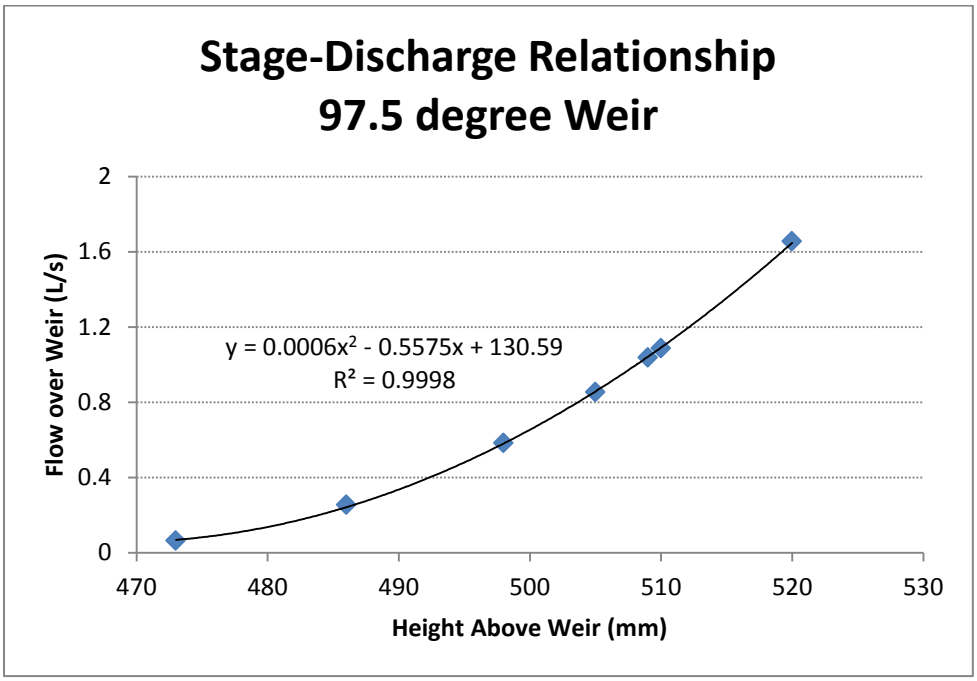


Figure D-1- Experimental calibration of 97.5 degree weir used in field rain garden monitoring.

Appendix E- Annual Precipitation Data

Table E-1- Average monthly precipitation data for Christchurch and Auckland, New Zealand (NIWA 2010).

	Average Monthly Precipitation											
Location	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Christchurch	41.7	39.2	53.7	54.1	56.3	66.2	78.7	69.1	47.3	52.8	44.2	49.4
Auckland	80.4	58.9	95.1	89.6	95	113.2	110.9	96	95.4	84.6	79.5	93.4

Note: Numbers represent 30 year average (1971-2000).

Table E-2- Average annual precipitation data for Christchurch and Auckland, New Zealand (NIWA 2010).

Location	**Average Annual Precipitation**
Christchurch	652.7
Auckland	1092

Note: Numbers represent 30 year average (1971-2000).